Mapping of Nutrient-Nitrogen Critical Loads for Selected National Parks in the Intermountain West and Great Lakes Regions

Natural Resource Technical Report NPS/ARD/NRTR—2014/895

Mapping of Nutrient-Nitrogen Critical Loads for Selected National Parks in the Intermountain West and Great Lakes Regions

T.J. Sullivan
T.C. McDonnell

E&S Environmental Chemistry, Inc.
P.O. Box 609
Corvallis, OR 97339

August 2014

The National Park Service, Natural Resource Stewardship and Science office in Fort Collins, Colorado, publishes a range of reports that address natural resource topics. These reports are of interest and applicability to a broad audience in the National Park Service and others in natural resource management, including scientists, conservation and environmental constituencies, and the public.

The Natural Resource Technical Report Series is used to disseminate results of scientific studies in the physical, biological, and social sciences for both the advancement of science and the achievement of the National Park Service mission. The series provides contributors with a forum for displaying comprehensive data that are often deleted from journals because of page limitations.

All manuscripts in the series receive the appropriate level of peer review to ensure that the information is scientifically credible, technically accurate, appropriately written for the intended audience, and designed and published in a professional manner.

This report received formal peer review by subject-matter experts who were not directly involved in the collection, analysis, or reporting of the data, and whose background and expertise put them on par technically and scientifically with the authors of the information.

Views, statements, findings, conclusions, recommendations, and data in this report do not necessarily reflect views and policies of the National Park Service, U.S. Department of the Interior. Mention of trade names or commercial products does not constitute endorsement or recommendation for use by the U.S. Government.

This report is available in digital format from the National Park Service Air Resources Division Website (http://www.nature.nps.gov/air/pubs/) and the Natural Resource Publications Management website (http://www.nature.nps.gov/publications/nrpm/). To receive this report in a format optimized for screen readers, please email irma@nps.gov.

Please cite this publication as:

Sullivan, T. J. and T.C. McDonnell. 2014. Mapping of nutrient-nitrogen critical loads for selected national parks in the intermountain west and great lakes regions. Natural Resource Technical Report NPS/ARD/NRTR—2014/895. National Park Service, Fort Collins, Colorado.

Contents

Maps

Maps (continued)

v

Maps (continued)

Maps (continued)

Tables

Executive Summary

Pardo et al. (2011a) compiled empirical data on nutrient-nitrogen (N) critical loads (CL) in ecoregions throughout the United States for a variety of nitrogen-sensitive resources, including lichens, mycorrhizal fungi, herbaceous plants and shrubs, and forests. These CL are generally reported by Pardo et al. (2011a) as ranges for a given ecoregion. Because ecosystems vary, sometimes substantially, within an ecoregion, the CL range for a given receptor may not apply to all plant communities or all portions of the ecoregion. When selecting appropriate CL for a specific national park it is important to examine the scientific basis of the CL, that is, whether the CL estimate was based on a specific vegetation type, and whether that vegetation type occurs in the park. Here, we evaluate empirical CL estimates for ecoregions and analyze available vegetation data to select the most appropriate CL range for selected national parks, based on the available data. We also examine N oxide emissions by county and total (oxidized and reduced) N deposition estimates in and near the parks. We compare deposition estimates for 2008 to park nutrient-N empirical CL to evaluate the likelihood that CL are exceeded in NPS areas. Our focus is on 12 national park units in the southwestern United States and the Northern Great Plains (NGPN) and the Great Lakes (GLKN) NPS Inventory and Monitoring (I&M) networks. The parks selected for study include many of those that have been identified as having N-sensitive ecosystems. Some of the parks have been experiencing in recent years increasing levels of nearby energy, agricultural, mineral extraction, and/or transportation development, with associated increasing N emissions.

The CL and exceedance estimates presented here were based on the lower limits of the ranges of CL, by ecoregion, reported by Pardo et al. (2011a). This was done to conform with the NPS mandate when considering CL to err on the side of resource protection.

Based on these lower limits of the CL ranges reported by Pardo et al. (2011a), terrestrial resources in most of the parks identified for study were either in exceedance of the nutrient-N CL or received ambient (year 2008) total wet plus dry N deposition that was below, but within 1 or 2 kg N/ha/yr of, the CL. Thus, nutrient sensitive terrestrial resources in some of these parks may be experiencing adverse impacts associated with CL exceedance. In other cases, appreciable increases in NO_x or ammonia (NH_3) emissions and deposition may trigger exceedances in the future.

Of the parks that were evaluated in this study, estimated nutrient-N CL exceedances were most pronounced in Voyageurs (VOYA), Mesa Verde (MEVE), Black Canyon of the Gunnison (BLCA), and Saguaro (SAGU) national parks. Large portions of Grand Canyon (GRCA), Arches (ARCH), Badlands (BADL), Theodore Roosevelt (THRO), and Wind Cave (WICA) national parks and Colorado (COLM) and Dinosaur (DINO) national monuments received N deposition in 2008 that was within 1 kg N/ha/yr or less below the nutrient-N CL. Much of Canyonlands National Park (CANY) received N deposition in 2008 that was below, but within 2 kg N/ha/yr of, the nutrient-N CL. These results provide estimates of relative risk of nutrient enrichment to sensitive vegetative receptors among the parks selected for study.

The emissions and deposition data shown in this report are estimates for 2008 and may not fully capture current emissions and deposition levels, particularly in areas where NO_x and/or NH_3

emissions have recently increased. Thus, it will be important to continue to track N emissions and deposition and to evaluate likely impacts on CL exceedance and associated impacts on nutrient-sensitive vegetation in these, and perhaps other, national park units.

Acknowledgments

Introduction

Atmospheric deposition of nutrient-N is an important ecosystem stressor in many national park units in the conterminous United States. The focus of this assessment is on park lands in the intermountain West and Great Lakes regions that are known to contain N-sensitive receptors and that may be exposed to increased N deposition in the future due to local energy, agricultural, transportation, and/or mineral extraction development. Sources of atmospheric N emissions are increasingly emerging as potentially important in these regions of the United States. Existing point and nonpoint N emissions sources associated with energy, transportation, and agricultural activities are common in many areas within these regions. Oil and gas exploration and production are ongoing and increasing in many areas. High-interest/visitation parks, potentially affected by these relatively new or increasing emissions sources, were identified by the NPS Air Resources Division (NPS-ARD) for detailed study (Table 1, Map 1). Most of the study parks are located in the southwest. Three are located in the Northern Great Plains Network (NGPN), and one in the Great Lakes Network (GLKN). Boundaries of desert regions relative to NPS network boundaries are shown in Map 2. Voyageurs NP, in the Great Lakes region, was included in the list of high-interest parks considered here because of concerns regarding emissions from potential taconite (a low-grade iron ore) extraction near the park.

Table 1. Parks in the Intermountain West, Northern Great Plains, and Great Lakes regions identified for this analysis as high-interest for evaluation of critical loads of nutrient-N deposition.

Park Code	Park Name	Network Code	NPS I&M Network Name	Ecoregion
ARCH	Arches NP	NCPN	Northern Colorado Plateau	North American Deserts
BADL	Badlands NP	NGPN	Northern Great Plains	Great Plains
BLCA	Black Canyon of the Gunnison NP	NCPN	Northern Colorado Plateau	North American Deserts
CANY	Canyonlands NP	NCPN	Northern Colorado Plateau	North American Deserts
COLM	Colorado NM	NCPN	Northern Colorado Plateau	North American Deserts
DINO	Dinosaur NM	NCPN	Northern Colorado Plateau	North American Deserts
GRCA	Grand Canyon NP	SCPN	Southern Colorado Plateau	North American Deserts
MEVE	Mesa Verde NP	SCPN	Southern Colorado Plateau	North American Deserts
SAGU	Saguaro NP	SODN	Sonoran Desert	North American Deserts
THRO	Theodore Roosevelt NP	NGPN	Northern Great Plains	Great Plains
VOYA	Voyageurs NP	GLKN	Great Lakes	Northern Forests
WICA	Wind Cave NP	NGPN	Northern Great Plains	Great Plains

Map 1. High-interest area for evaluation of nutrient-N CL issues in the intermountain West and Great Lakes regions, showing park lands selected for intensive study. Voyageurs National Park, in the Great Lakes Network, is included because of concerns regarding NO_x emissions from taconite mining near the park. The location of Joshua Tree National Park, although outside the circumscribed high-interest area, is also shown because it is the source area for considerable data regarding the response of desert plant communities to N input.

Map 2. Location of desert regions in the southwestern United States and northern Mexico relative to NPS network boundaries.

Emissions of NO_x and NH_3 from these and other human activities, with resulting increases in N deposition, have the potential to affect soils, vegetation, and surface waters in national parks and monuments located generally downwind of emissions sources. In particular, increased N loading to natural ecosystems can cause changes in plant communities, sometimes favoring exotic and invasive species over native species.

Emissions of NO_x and other pollutants from oil and gas development are rapidly emerging as an important threat to air quality in various regions of the United States. Recent and ongoing oil and gas development in the United States is accomplished mainly by hydraulic fracturing (fracking). The process of fracking involves 1) clearing a well site, 2) drilling a hole many hundreds or thousands of meters to reach the location of a targeted shale deposit, 3) rotating the drilling apparatus to bore a long horizontal pathway through the shale, 4) fracturing the shale under pressure from water and steam, and 5) capturing the natural gas or oil that flows from the wellhead. Emissions of NO_x from traditional vertical drilling and from other vehicular and industrial sources also impact air quality and pollutant deposition in NPS parks. Emissions of reduced N from confined animal feeding operations (CAFOs) and other agricultural developments are important emissions sources in some regions.

Oil and gas shale basins are widely distributed across many parts of the United States. One region where such shale formations predominate runs in a band from the Canadian border north of Theodore Roosevelt NP (THRO) to the desert southwest in and around Canyonlands National Park (CANY; National Parks Conservation Association [NPCA] 2013). Many thousands of new wells have been installed, most since about 2008, and the installation rate is expected to continue to increase (http://cogcc.state.co.us/Home/gismain.cfm; NPCA 2013). The area around Theodore Roosevelt NP is a major center of oil and gas development. The NPCA (2013) reported an increase in the number of active wells in the vicinity of Theodore Roosevelt NP from about 1,000 in 2008 to over 6,000 in 2012 (North Dakota Industrial Commission 2012; http://www.dmr.nd.gov/oilgas/maps/2012). The Uintah Basin of Utah, including the area adjacent to Dinosaur NM, is another center of recent oil and gas development with thousands of wells (http://gis.utah.gov/data/energy/oil-gas/).

Emissions of NO_x from an individual well, CAFO, or industrial source may be insignificant to the broader landscape. However, the cumulative effects of many sources may be substantial. Because the oil and gas source type is increasing rapidly, few data are available with which to assess the magnitude of the effects on N deposition, ozone (O_3) formation, or other air or water pollution issues. There is an accelerating need to compile and evaluate data on emissions, deposition, and effects, especially in and in proximity to Theodore Roosevelt NP, Dinosaur NM, and several other parks.

The CL is the pollutant loading rate below which adverse impacts are not expected to occur to sensitive ecosystem receptors. The CL is commonly calculated to protect against either nutrient-N enrichment or acidification. Resources differ in their sensitivity to damage from air pollutant deposition of N and/or S. The CL are variable by ecoregion and vegetation type. In terrestrial systems, the CL are often especially low (high sensitivity to damage) for protection of lichens and mycorrhizal fungi. In addition, there are multiple approaches that can be used to estimate the CL. These include deposition gradient studies, experimental N and/or S addition, and steady-state or dynamic modeling. Each approach entails strengths and weaknesses. Thus, an individual national

park will not have a single CL. Rather, it will have a matrix of CL depending on the resources to be protected, the method used to estimate the CL, and the desired level of protection. Available estimates of CL for a particular ecosystem element are constantly changing as additional studies are conducted and new information becomes available. Therefore, the CL estimates presented in this report are subject to change, and likely will change, in the future.

In 2011, Pardo et al. (2011a) summarized empirical N CL for various Level 1 ecoregions of the United States (CEC 1997). Individual chapters of the Pardo et al. (2011a) compilation address land areas evaluated in this report. These chapters were written by Allen and Geiser (2011), Clark (2011), and Pardo et al. (2011b). These CL define the amount of N deposition below which ecosystems or specific ecosystem receptors are unlikely to be harmed by nutrient enrichment processes. Thus, the empirical CL compiled by Pardo et al. (2011a) identified tipping points for sensitive resource damage due to N enrichment under elevated N input levels. This report uses the CL data of Pardo et al. (2011a) together with emissions and deposition estimates described in Appendix A to provide an assessment of potential impacts of NO_x emissions on sensitive ecosystem receptors for the high interest parks listed in Table 1.

The focus of this report is on effects related to nutrient deposition. Increased loading of N, which is frequently limiting to plant growth, benefits some species at the expense of others. Under high N loading, plant species richness and biodiversity often decrease. Non-native invasive species are often among those that benefit the most from N addition. In arid and semi-arid plant communities, there can also be interactions with water availability.

Pardo et al. (2011a) reported nutrient CL from numerous studies, aggregated to an ecoregion scale. Here, we build upon that effort by developing park-specific estimates of CL. These estimates are based on park vegetation data and more recent estimates of atmospheric N deposition. It is hoped that results of these analyses will improve understanding of CL and exceedance at the park scale within the study area addressed in this report. Refined information on CL and air quality related values (AQRV) within specific national park units is needed to address ongoing and new threats to park resources from oil and gas development, agricultural activities, vehicular emissions, mineral extraction, and industrial development.

Approach

Vegetation Typing

The U.S. Geological Survey (USGS) constructed vegetation maps for most of the national parks selected for inclusion in this study as part of the National Park Service's (NPS) Inventory and Monitoring (I&M) Program. The I&M Program and the USGS classified, described, and mapped vegetation in 250 national parks from photographic interpretation of 1:12,000-scale color infrared aerial photos taken in 1997. Vegetation map classes (0.5 ha minimum mapping unit) were determined through field reconnaissance, data collection, and analysis in accordance with the National Vegetation Classification System (FGDC 1996). These vegetation mapping unit data were used in this analysis to determine the spatial distribution of CL values within each of the high-interest parks.

Recent I&M vegetation maps were not available for one study park (Saguaro NP) at the time of preparing this report. For that park, vegetation types and their spatial distribution were obtained from the LANDFIRE (2008) database. The LANDFIRE existing vegetation type (EVT) data were used as the basis for vegetation classification for that park. The EVT data were developed at 30 m resolution using statistical decision tree models based on Landsat imagery, elevation and biophysical landscape characteristics. The data were intended to represent the current vegetation species composition for a given site.

The vegetation community types associated with the map units for each vegetation dataset were grouped according to the sensitive ecosystem components listed in Pardo et al. (2011a) for each ecoregion. The original vegetation community name often listed the dominant species such as chokecherry shrubland or ponderosa pine woodland. These vegetation map units were classified as "shrubland" and "woodland" ecosystem types, respectively, to match the Pardo et al. (2011a) ecosystem component designations. Differences between the dominant species represented in the studies compiled by Pardo et al. (2011a) and the species included in the park-specific vegetation datasets were not considered. Vegetation survey metadata, which provided additional information on the species present within a given map unit, were consulted to ensure proper classification into the Pardo et al. (2011a) ecosystem components.

Results of analyses conducted for this study are organized hierarchically in this report, by ecoregion (Northern Forests, Great Plains, and North American Deserts), by NPS I&M networks within ecoregions, and by national park units within networks. Within each subject network and national park unit, we report available data on empirically determined nutrient-N CL values, NO_x emissions as updated from the NEI by WestJump, total N (including both oxidized and reduced chemical species) deposition, and CL exceedances.

Critical load estimates assigned to various general vegetation types by Pardo et al. (2011a) were extracted from their report and matched with parks in which these vegetation types occurred. These CL estimates were specific to a variety of vegetation types, and included (depending on data availability) lichens; mycorrhizal fungi; and various tree, shrub, forb, and graminoid communities.

Critical Loads

National parks and national park networks considered in this analysis were classified according to the Level 1 ecoregions in which they occurred (Table 1). Critical load estimates compiled by Pardo et al. (2011a) for Level I ecoregions were matched to the high-interest parks in each ecoregion. Critical loads were mapped for the identified high-interest parks at the scale of available vegetation and deposition data to reflect within-park differences in the sensitivity of park resources. For evaluating CL, we rely upon Pardo et al.'s (2011a) lower end of reported empirical CL. This was done in order to comply with the NPS mandate to err on the side of resource protection in evaluating CL and exceedance. Information on CL from sources other than Pardo et al. (2011a) was used to evaluate the sensitivity of plant communities to N input. Such information was evaluated together with CL estimates compiled by Pardo et al. (2011a) to evaluate the state of scientific understanding regarding park resource sensitivity within the study area to N inputs.

We applied Pardo et al.'s (2011a) compilation of ecoregional CL estimates to produce CL maps of the selected national parks and monuments considered to be most at risk from increases in N deposition from oil and gas development, mineral extraction-related activities, agricultural, and other emissions sources. Using the available spatial vegetation type data described above, CL estimates compiled by Pardo et al. (2011a) were mapped across national parks for a variety of plant community endpoints and receptors. The CL were analyzed with respect to vascular plant species in this report. The Pardo et al. (2011) report also includes CL for lichens, bryophytes and mycorrhizae for some ecoregions. However, these organisms are not represented in the vegetation datasets used in the analyses reported here.

Emissions and Deposition

Emissions of NO_x and NH_3 from multiple emissions source types were compiled for this study from available emissions inventory data for 2008 (Appendix A). The West-Wide Jump Start Air Quality Modeling Study (WestJumpAQMS) compiled estimates of (NO_x) N oxide emissions from oil and gas exploration and production using data from the U.S. Environmental Protection Agency's (EPA) National Emissions Inventory (NEI) and other sources, and of total wet plus dry N deposition using the Comprehensive Air Quality Model with Extensions (CAMx) model for the year 2008. The NEI estimates were updated to include emissions sources of NO_x, but not NH_3 by WestJump.

Total (wet + dry) atmospheric N deposition estimates developed for the year 2008 in the WestJumpAQMS were used as the basis for evaluating CL exceedance. These estimates included both oxidized (NO_x) and reduced (NH_4^+ and NH_3) atmospheric deposition. The CAMx model was used to develop deposition estimates at 12 km resolution for the western states, including the WestJump AQMS basins. The map resolution for other portions of the United States was 36 km. Such coarse resolution smoothes out fine-scale differences related to orography in mountainous areas.

Results from the WestJumpAQMS reported here relied on several emissions models and one photochemical grid model to produce estimates of total N deposition. The emissions models included:

SMOKE (Sparse Matrix Operator Kernel Emissions): generates hourly gridded speciated emissions from mobile, non-road, area, point, fire and biogenic emission sources,

MOVES (Motor Vehicle Emission Simulator): generates emissions inventories and rate lookup tables for on-road mobile sources and is used to provide input to SMOKE,

MEGAN (Model of Emissions of Gases and Aerosols in Nature): generates net emissions estimates from terrestrial ecosystems.

Data sources used to develop the emissions estimates with these models are described in detail in the WestJumpAQMS Draft Final Modeling Protocol (ENVIRON International Corporation 2012). Each of these emissions models was used to develop estimates of N emissions as input into a photochemical grid model. Two photochemical grid models, CAMx (Comprehensive Air Quality Model with Extensions) and CMAQ (Community Multiscale Air Quality), were scheduled to be implemented to develop estimates of N deposition in the WestJump study. However, only CAMx results were available for use in analyses presented in this report. Other estimates of wet and dry deposition are also available from a variety of sources, including combinations of approaches, such as the recent TDEP effort. Comparison of results of such efforts is beyond the scope of this analysis.

Further details regarding deposition modeling procedures are included in the WestJumpAQMS Draft Final Modeling Protocol (ENVIRON International Corporation 2012; Chapter 6). Total NO_x emissions and total N deposition estimated by WestJump for the conterminous United States are summarized in Appendix B.

Nitrogen oxide emissions are reported here for each network as a series of four maps. Each depicts a component of the NO_x emissions, aggregated by county. The first map in each series gives the county-level estimated oil and gas development NO_x emissions for 2008 in units of tpy. Within the WestJump study domain, emissions estimates represented on this map are based on existing NEI data, as augmented by the WestJump effort; elsewhere, emissions estimates only reflect NEI data with no additional data compilation by WestJump. County-level emissions only tell part of the story, however. Counties differ in size, especially between the eastern and western portions of the country. Therefore, the second map in each series shows county-level emissions, normalized by county area (as tons per square mile per year [tons/mi2/yr]). The third map shows the estimated total NO_x emissions from all sources, including oil and gas exploration and production. The fourth and final map depicts oil and gas development NO_x emissions as a percentage of the total emissions from all sources. In many counties within the intermountain West, oil and gas exploration and production accounts for more than half of the total NO_x emissions, based on 2008 data. This percentage may be higher now in some areas due to the rapid expansion in oil and gas exploration and production since 2008

Exceedance

Areas where CL are exceeded by current deposition were identified, using maps of total N deposition developed by adding CAMx model estimates of dry and wet N deposition. We also mapped the difference between CL and ambient deposition to show how close ambient deposition is to the CL and to indicate where CL are being exceeded.

The resulting data and maps will assist NPS in responding more quickly and accurately to proposed development near parks and in assessing natural resource conditions. This information will be helpful in quantifying how much deposition is likely to harm resources in the context of existing sensitive receptors, available dose/response data, and current and future estimates of emissions and deposition.

Results

Critical Load

North American Deserts Ecoregion

Woodland, shrubland, and grassland vegetation types in the North American Deserts Ecoregion are known to be highly sensitive to N inputs. Allen and Geiser (2011) estimated that the lower end of the range of nutrient N empirical CL for the plant communities in this ecoregion is about 3 kg N/ha/yr. National parks in this ecoregion received estimated N deposition in 2008 that was near, and in many locations above, that level. Thus, NPS lands are in many cases in exceedance, or close to being in exceedance, of this conservative estimate of the CL for resource protection.

The North American Deserts Level I ecoregion includes the Chihuahuan, Mojave, and Sonoran deserts and national park lands within the Southern Colorado Plateau (SCPN), Northern Colorado Plateau (NCPN), Sonoran Desert (SODN), and Mojave Desert (MOJN) networks (Maps 1 and 2). Within these largely desert networks, there are eight national park lands considered in this analysis: Arches NP, Black Canyon of the Gunnison NP, Canyonlands NP, Colorado NM, Dinosaur NM, Grand Canyon NP, Mesa Verde NP, and Saguaro NP. In addition, Joshua Tree National Park (JOTR), although not explicitly considered for this analysis, is in proximity to the study region and has been relatively well-studied regarding ecological effects of atmospheric N deposition.

Responses of desert vegetation communities to N addition have been shown to include change in plant species composition, increase in biomass of non-native invasive species, and decrease in relative abundance and richness of native herbaceous species (Allen and Geiser 2011). The abundance of invasive non-native plant species has increased during recent decades in desert ecosystems, likely due in part to N deposition (Brooks 2003, Allen et al. 2009). Exotic grasses, in particular, respond strongly to increased N supply in desert environments. Increased grass cover, in turn, can contribute to greater fire frequency (Brooks et al. 2004, Brooks and Matchett 2006).

Critical loads of nutrient-N deposition have been estimated in a number of studies for vegetative communities in the North American Deserts ecoregion, mostly in the southern portion. These estimates may be applicable for protection of sensitive receptors in a number of national park desert networks, including the Southern Colorado Plateau, Northern Colorado Plateau, and Sonoran Desert networks.

Biomass of native blue grama grass in desert grassland at the Sevilleta Long-Term Ecological Research (LTER) site in New Mexico increased in response to N addition at a rate of 20 kg N/ha/yr (Báez et al. 2007). In contrast, a change in ambient N deposition at the control plot during the 16-year experiment from 1.7 to 2.4 kg N/ha/yr did not cause a change in blue grama biomass. Based on this work, the CL to protect blue grama biomass may be somewhere between 2.4 and 20 kg N/ha/yr.

Experimental N addition has been shown to reduce soil moisture in sagebrush steppe. This would be expected to decrease herbaceous plant productivity (Inouye 2006). Experimental fertilization with N at very high levels (72 kg N/ha/yr) at the Jornada LTER site in New Mexico caused increased growth of native annual forbs during winter, but decreases during summer (Gutierrez and Whitford 1987).

This variable response may have been due, at least in part, to increased water limitation during summer. Barker et al. (2006) determined that water availability was the most important limiting factor for productivity of the desert shrub creosote bush (Larrea tridenta).

Nitrogen fertilization studies in the Sevilleta grasslands of the Chihuahuan Desert documented changes in desert grassland species composition and biomass at high levels of N input (22 kg N/ha/yr) after one year. No changes were observed under ambient deposition (2.4 kg N/ha/yr). Thus, results of this study suggested that the CL to protect these grassland species may be higher than 2.4 kg N/ha/yr, but the uncertainty regarding the actual CL level is substantial.

Allen et al. (2009) estimated the CL to be approximately 8.4 kg N/ha/yr (3.4 kg N/ha/yr deposited plus 5 kg N/ha/yr fertilized) during wet years at JOTR in the Mojave Desert, just outside the Southern Colorado Plateau Network, to protect against increased coverage of invasive non-native grasses. They found that the response of desert vegetation was determined by multiple factors in addition to the N deposition level, also including soil properties and moisture availability. These findings were further corroborated by DayCent modeling conducted by Rao et al. (2010). Modeling results suggested the following CL levels:

Lower elevation desert dominated by the invasive common Mediterranean grass (Schismus arbatus)	3.1-8.2 kg N/ha/yr
Higher elevation desert dominated by the invasive grass red brome (Bromus rubens)	3.0-5.7 kg N/ha/yr

There were important connections with fire risk. Modeled deposition above 8.2 kg N/ha/yr (low elevation) and 5.7 kg N/ha/yr (high elevation), thus above the respective CL at these two locations, caused fire risk to be controlled by precipitation rather than N input (Rao et al. 2010). The desert environment in and near JOTR appears to be experiencing increasing impacts associated with changes in the grass fire cycle (Brooks and Matchett 2006). These data help to inform dose-response and CL issues elsewhere in the North American Deserts Ecoregion.

From 1989 to 2004, Baez et al. (2007) observed a 43% increase in ambient N deposition at a study site in the Chihuahuan desert, from 1.7 to 2.5 kg N/ha/yr, resulting in an additional 5.9 kg N/ha deposition over that time period. This increase in N deposition may have caused changes in the plant community, as suggested by fertilization studies of blue grama and black grama (Bouteloua eriopoda) with addition of relatively large amounts of N over a short period of time. With additions of 20 kg N/ha in one season, blue grama was favored over black grama, the current dominant species (Báez et al. 2007).

Enhanced N could play a role in the recently observed invasion of some exotic plant species within parts of the Mojave and Sonoran deserts. However, water is often more limiting than N in desert ecosystems. Brooks (2003) found that plant responses were influenced by specific rainfall events rather than by average annual rainfall, with the annual plants thriving in a year when high rainfall events triggered germination. In the Mojave desert, the shrub creosote bush showed no increased growth response to experimental N additions at 10 and 40 kg N/ha/yr as calcium nitrate ($Ca(NO_3)_2$),

but did respond to increased water (Barker et al. 2006). Conversely, invasive annuals showed a greater response to elevated N than did native species. Though their invasion was correlated with greater N deposition, no causation has been established (Fenn et al. 2003).

In the Chihuahuan desert, greater ecosystem N losses in shrublands (0.33 kg/ha/yr) versus grasslands (0.15 kg/ha/yr) corresponded with greater water runoff in shrublands. Even with runoff nutrient losses, both vegetation types showed a net accumulation of total N, just over 2 kg N/ha/yr, with atmospheric N deposition of about 2.5 kg/ha/yr (Schlesinger et al. 2000).

Fertilization experiments by Brooks (2003) in the Mojave Desert showed that increased levels of N deposition could favor the expansion of non-native species where they are already prevalent (Brooks 2003). At very high N application rates of 32 kg N/ha/yr over two years, both density and biomass of non-native plants increased. Non-native biomass increased 54% while native species biomass declined by about 39%.

Allen et al. (2009) evaluated the effects of reactive N on native and non-native plant species in JOTR. This park receives moderate levels of air pollution, including N, from the Los Angeles Basin. Extractable soil N was generally higher at study sites that received higher atmospheric N deposition. Nitrogen fertilizer was applied at levels of 5 and 30 kg N/ha/yr at four sites over a two-year period. Low-elevation sites were dominated by creosote bush scrub and higher-elevation sites by pinyon-juniper woodland. Non-native grass biomass increased significantly at three of four treatment sites that received 30 kg N/ha/yr, but not at the sites that received 5 kg N/ha/yr. Experimental studies by Allen et al. (2009) at JOTR suggested that the critical load of N to protect desert vegetation in this park against increased invasive grass biomass due to N enrichment during wet years may be as low as about 8 kg N/ha/yr (Allen and Geiser 2011). Modeling results using the daily Century model (DayCent; Brooks and Matchett 2006, Allen and Geiser 2011) were similar.

Application of N fertilizer along an N depositional gradient in creosote bush scrub and pinyon-juniper woodland in JOTR suggested that the response of desert vegetation to added N varied by wetness condition. Fertilization rates ranged from 5 to 30 kg N/ha/yr, in addition to about 3 to 12 kg N/ha/yr of ambient atmospheric deposition (Tonnesen et al. 2007, Allen and Geiser 2011). During 2003, which was a dry year, vegetation biomass remained unchanged. During 2004, a moderate precipitation year, two of four experimental plots showed increased biomass of non-native invasive grasses at addition of 30 kg N/ha/yr, but not at 5 kg N/ha/yr. During the wettest year (2005), invasive grass biomass increased at both N treatment levels (Allen et al. 2009). Productivity of native forbs decreased in response to N addition at these sites, but increased at another treatment site that had only sparse invasive grass cover.

Creosote bush demonstrated a twofold to threefold increase in stem growth, a 2.5- to 4-fold increase in viable seed production, and a 17% to 35% increase in leaf N in response to large N additions at sites around Mono Lake, CA. The experimental N was applied in March and November as NH_4NO_3, at a high cumulative addition rate of 233.6 g N per plant (Drenovsky and Richards 2005).

The DayCent model was applied to plant communities in the deserts of southern California by Rao et al. (2009, 2010). Invasive non-native grass production was simulated from 2003 to 2008 under varying levels of N input and precipitation, resulting in changing fire frequency. Simulated fire risk increased above N deposition of 3 kg N/ha/yr. The simulated risk leveled off at N deposition above 5.7 kg N/ha/yr in pinyon-juniper and above 8.2 kg N/ha/yr in creosote bush scrub (Rao et al. 2010). Earlier studies (Brooks 2003, Brooks and Matchett 2006) had suggested a link between fire risk and increased grass productivity caused by elevated atmospheric N deposition.

Allen and Geiser (2011) reported CL for the North American Deserts ecoregion. Values of nutrient-N CL were reported for lichens and for several vascular plant communities as follows:

Lichens. Change in community composition; increased N concentration in thallus	3 kg N/ha/yr	Porter (2007), Geiser et al. (2008)
Vascular plants. Change in community composition of shrubland, woodland, and desert grassland	3-8.4 kg N/ha/yr	Allen et al. (2009), Inouye (2006), Rao et al. (2010)

Estimates of CL in portions of the ecoregion that include this network were considered to be fairly reliable for protecting vascular plants, but less reliable (mainly expert judgment) for protecting lichens (Allen and Geiser 2011).

Southern Colorado Plateau Network
SCPN contains two parks considered in this analysis: Grand Canyon NP and Mesa Verde NP. Critical loads of nutrient-N deposition are analyzed for each of these parks in the sections that follow.

Does Response and Critical Loads
Grand Canyon National Park

Most of Grand Canyon NP is located within the North American Deserts Level I ecoregion, with the western portion of the park within the Mojave Desert Level III ecoregion. Although a minor portion of the park occurs within the Temperate Sierras Level I ecoregion, there is limited information available on CL for the Temperate Sierras and what information is available pertains mostly to NO_3-leaching in Mexico. Our analyses for Grand Canyon NP presented here are therefore focused on the North American Deserts ecoregion which covers most of the park.

The Mojave is the driest and hottest of the North American deserts. Therefore, scarcity of water frequently limits plant growth. Creosote bush and Joshua tree (Yucca brevifolia) are characteristic plants of the Mojave Desert.

Portions of Grand Canyon NP covered by woodland, shrubland, and grassland are shown in Map 3. The majority of the park is covered by shrubland. Each of these plant community types was estimated by Allen and Geiser (2011) to be sensitive to nutrient-N deposition and to have CL in the range of 3 to 8.4 kg N/ha/yr. Locations in Grand Canyon NP where these plant communities occur

and where ambient deposition is higher than 3 kg N/ha/yr are in exceedance of the CL for protecting these potentially N-sensitive plant communities against change in community composition. This same CL (3 kg N/ha/yr) was also judged by Allen and Geiser (2011) to be applicable to protection of lichen community composition in desert plant communities such as are found in Grand Canyon NP.
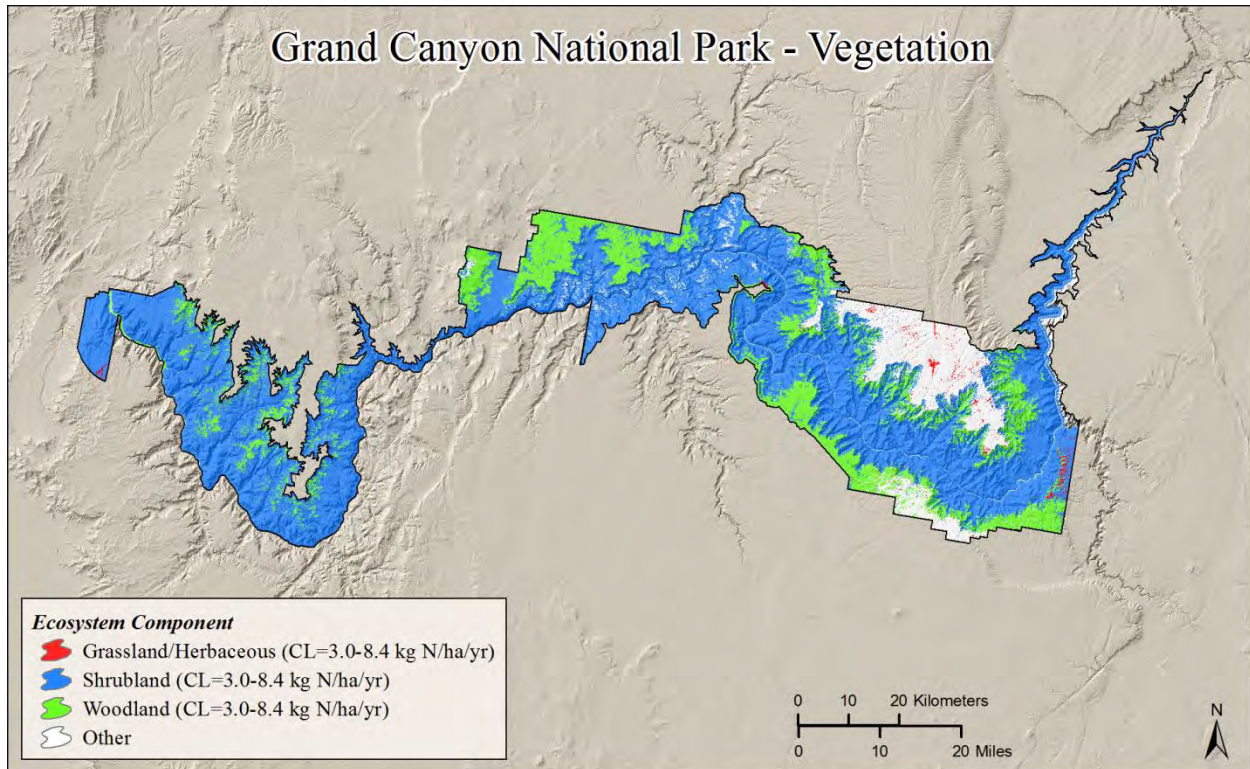
Map 3. Locations in Grand Canyon National Park having vegetation types characterized by the presence of plant communities expected to be most sensitive to nutrient-N deposition, based on nutrient critical load values compiled by Allen and Geiser (2011).

Mesa Verde National Park

Portions of Mesa Verde NP covered by woodland, shrubland, and grassland are shown in Map 4. Shrublands predominate. Each of these plant community types was estimated by Allen and Geiser (2011) to be sensitive to nutrient-N deposition and to have CL in the range of 3 to 8.4 kg N/ha/yr. Locations in Mesa Verde NP where these plant communities occur and where ambient deposition is higher than 3 kg N/ha/yr are in exceedance of the CL for protecting these potentially N-sensitive plant communities against change in community composition. This same CL (3 kg N/ha/yr) was also judged by Allen and Geiser (2011) to be applicable to protection of lichen community composition in these desert plant communities such as are found in Mesa Verde NP.
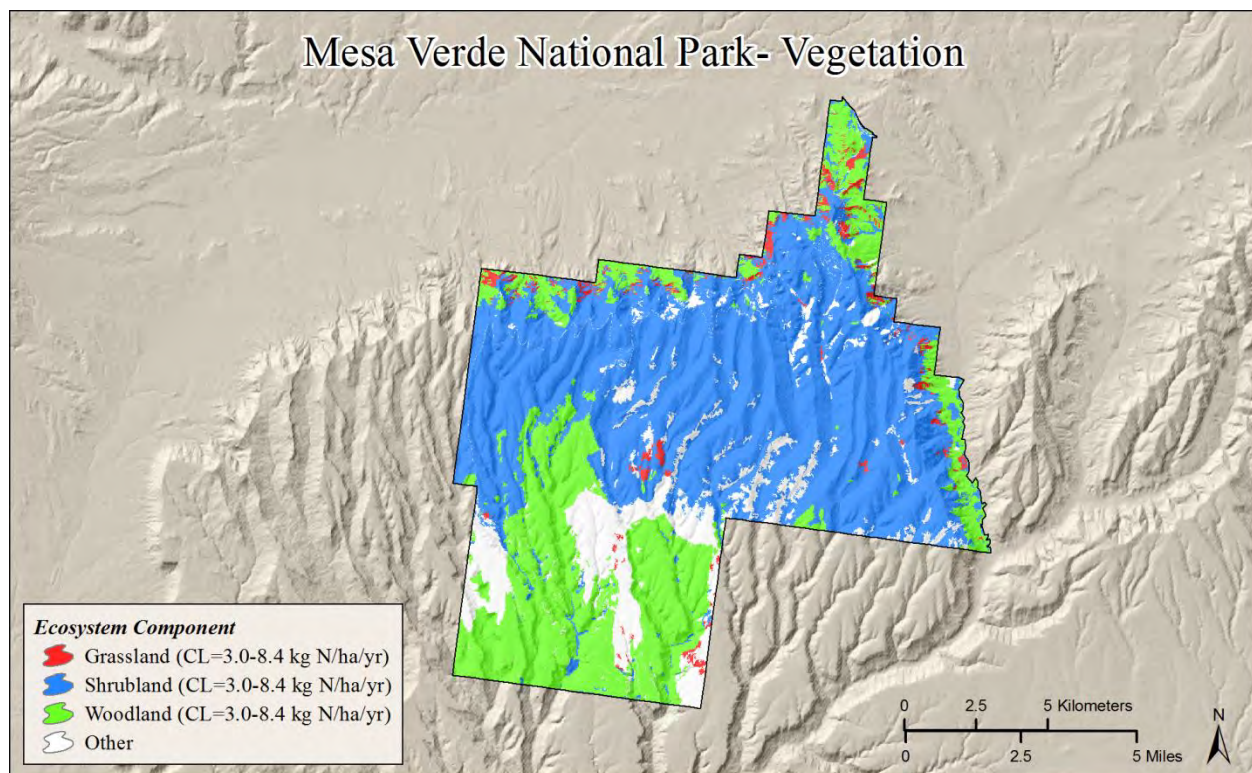
Map 4. Locations in Mesa Verde National Park having vegetation types characterized by the presence of plant communities expected to be most sensitive to nutrient-N deposition, based on nutrient critical load values compiled by Allen and Geiser (2011).

Emissions and Deposition

WestJump emissions data have been compiled for the northwestern portion of the SCPN, near Mesa Verde NP, but not in the vicinity of Grand Canyon NP. Some counties near Mesa Verde NP showed relatively high oil and gas NO_x emissions in 2008, in the range of 5,000 to 25,000 tpy (Map 5); these were mostly large counties, and therefore the emissions per unit area were generally not so pronounced (Map 6). Near Grand Canyon NP, the total NO_x emissions estimates depicted on Map 7 only include NEI sources without additional WestJump data.
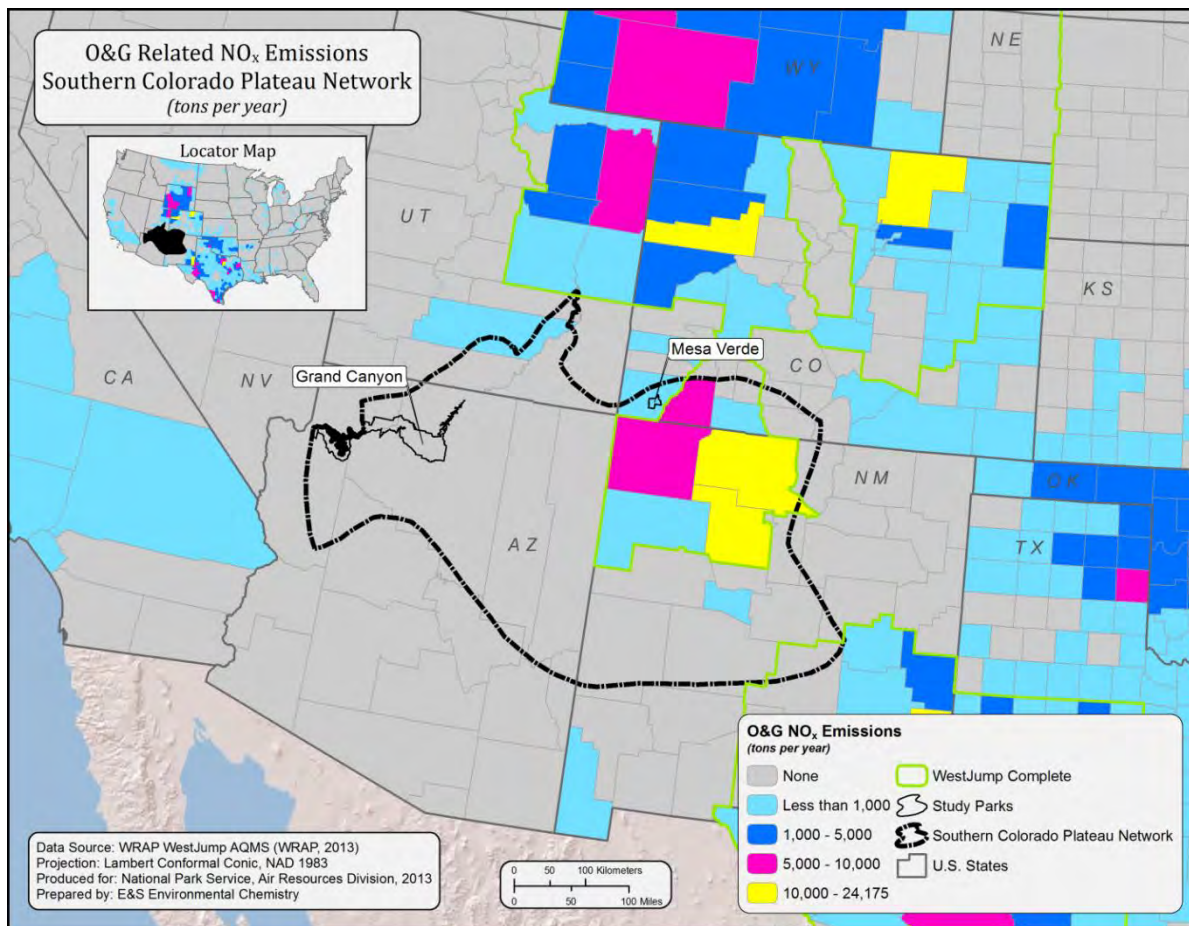
Map 5. Estimated oil and gas NO_x emissions (tpy), by county in the Southern Colorado Plateau Network for the year 2008, which is the most recent year for which data are available at the time of this writing. Note that oil and gas emissions levels in some areas have likely increased substantially in more recent years.
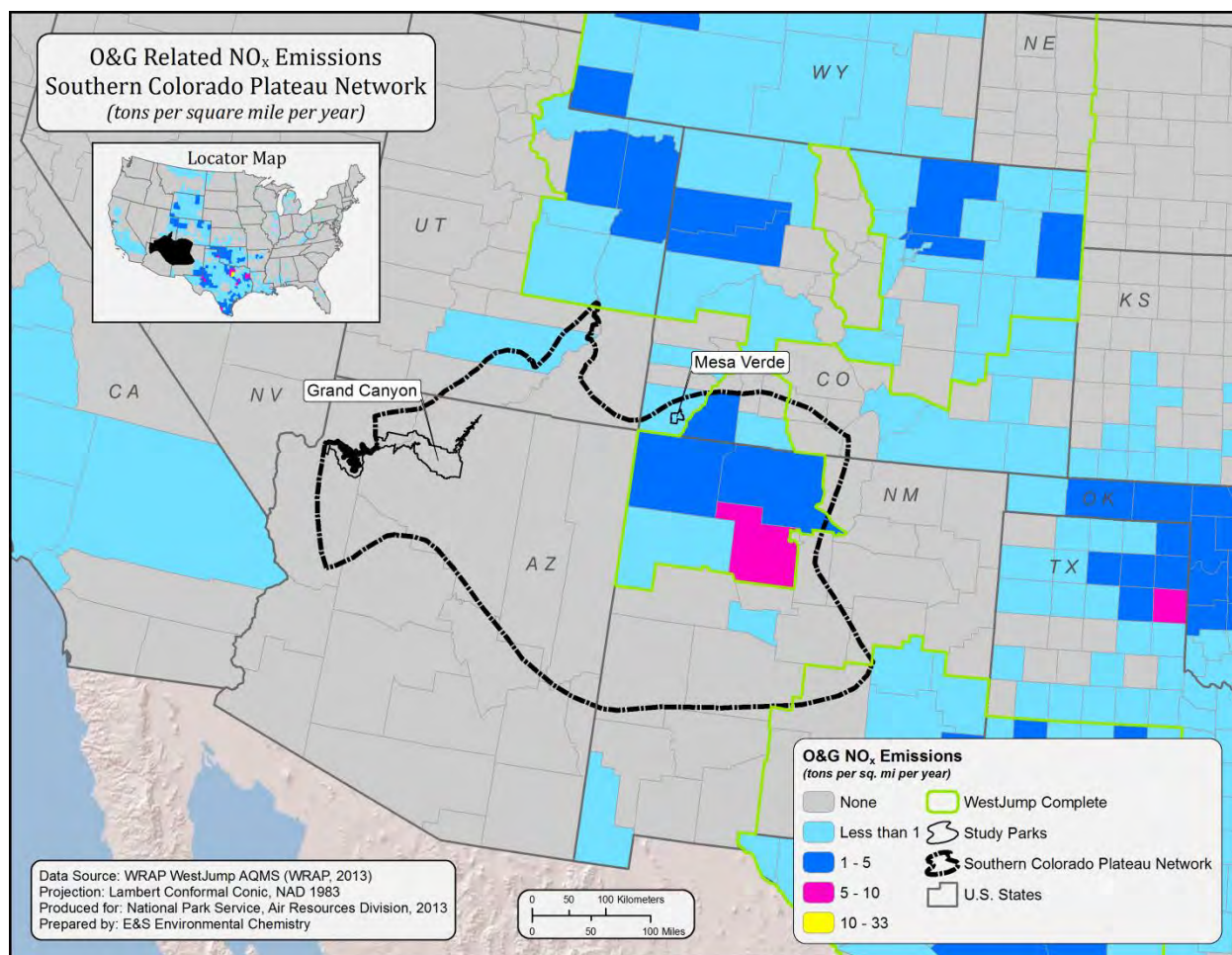
Map 6. Estimated oil and gas NO_x emissions per unit land area (tons/mi^2/yr) in the Southern Colorado Plateau Network for the year 2008.

There may be additional small NO_x sources that are not included on these maps. Total NO_x emissions from all source types are relatively high (10,000-25,000 tpy) near Grand Canyon NP, but somewhat lower in the immediate vicinity of Mesa Verde NP. Nevertheless, total NO_x emissions south of Mesa Verde NP are higher, in the range of 25,000 to 50,000 tpy (Map 7). Just south of Mesa Verde NP, oil and gas NO_x emissions constitute an estimated 20-40% of NO_x emissions from all source types. To the east and southeast of Mesa Verde NP, that percentage is even higher (> 60%; Map 8). The highest overall NO_x emissions in and near the network boundary occurred near Grand Canyon NP (Map 7), but the proportional contribution of oil and gas development to these total NO_x emissions appeared to be higher near Mesa Verde NP (Map 8). The Navajo Generating Station near Grand Canyon NP and the San Juan and the Four Corners Generating Stations in NW New Mexico near Mesa Verde NP are among the largest electric utility sources of NO_x emissions in the U.S. Estimates of total wet plus dry N deposition for SCPN, generated with the CAMx model, were highly variable. They ranged from less than 3 kg N/ha/yr in and around Grand Canyon NP to the range of 2-6 kg N/ha/yr in and around Mesa Verde NP (Map 9).
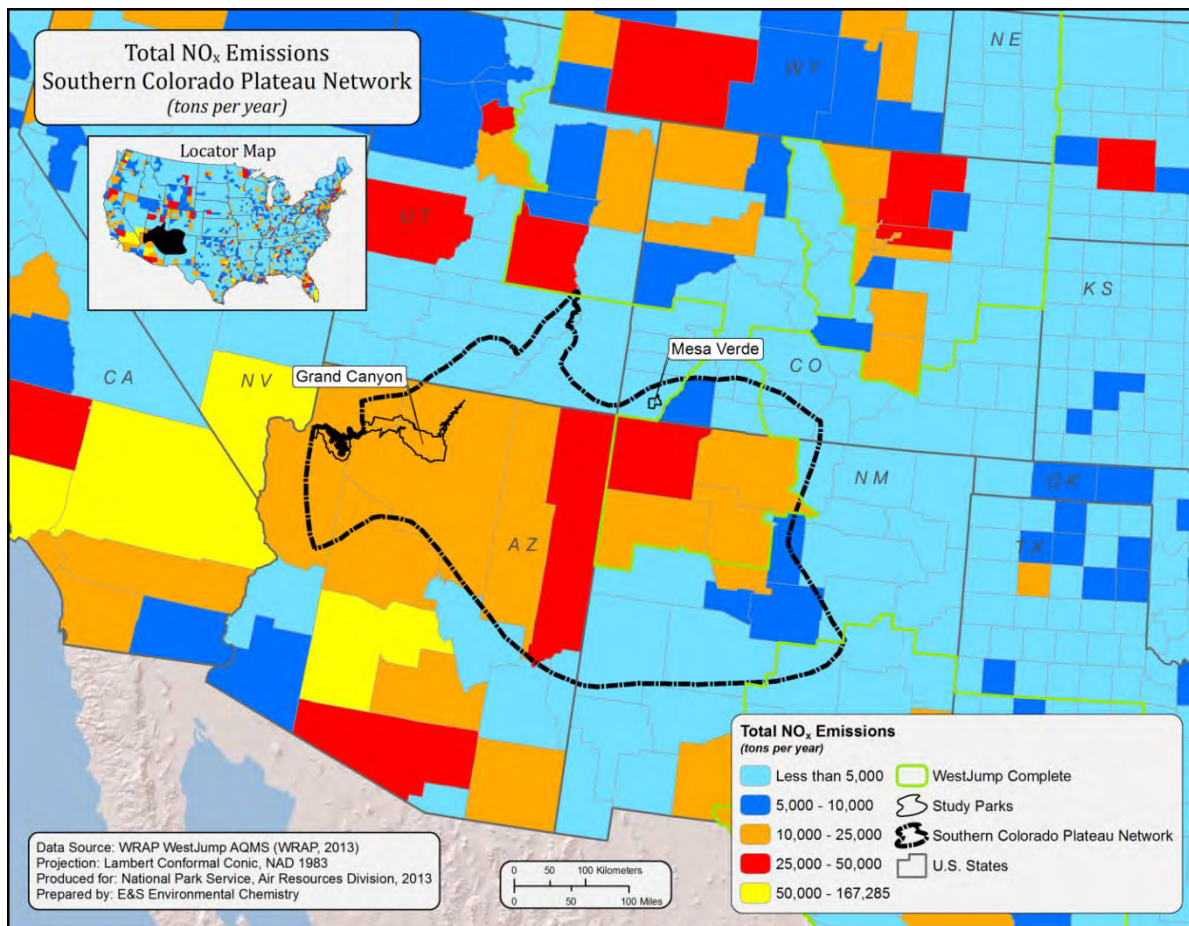
Map 7. Estimated NO$_x$ emissions from all sources (tpy), including oil and gas development, in the Southern Colorado Plateau Network for the year 2008.
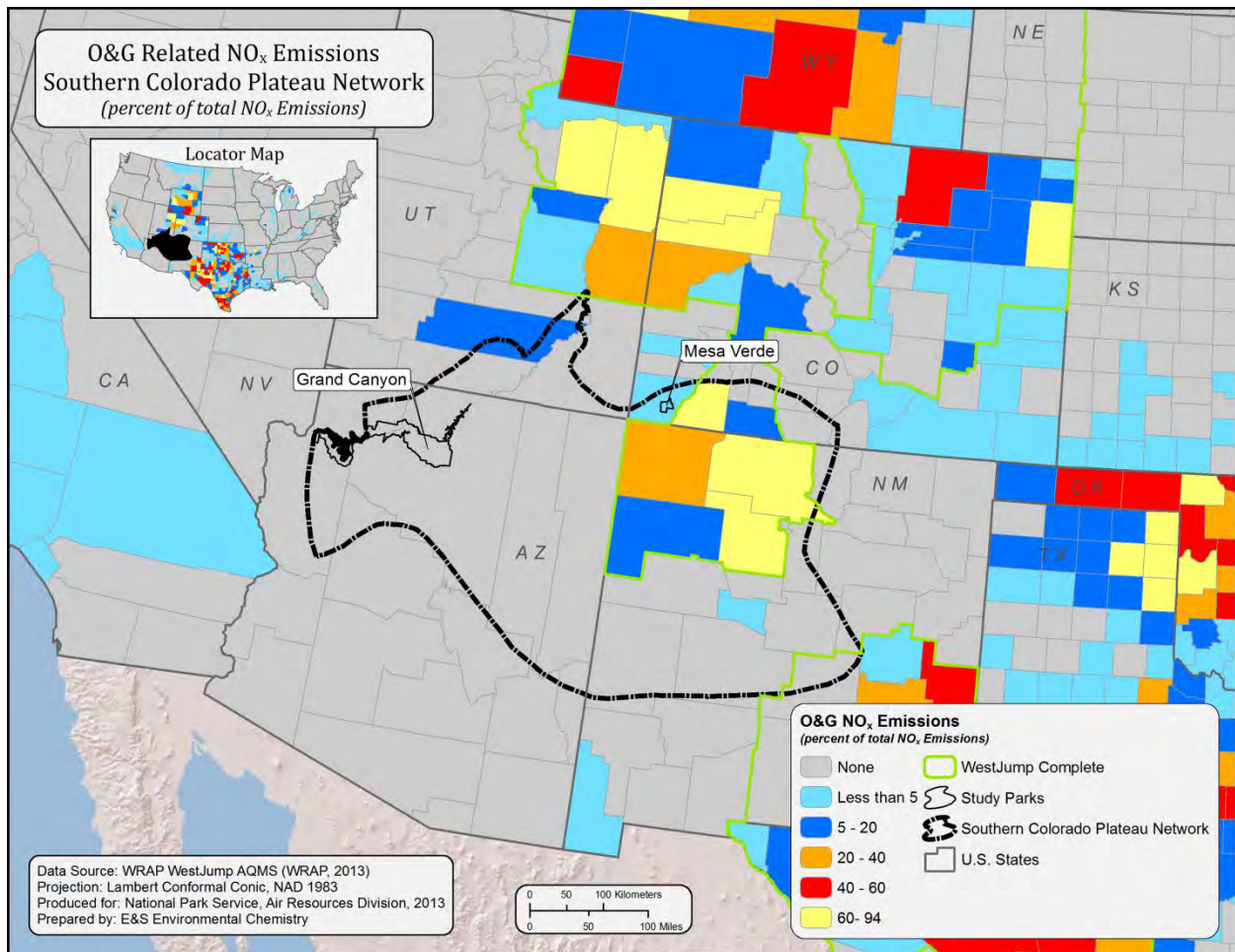
Map 8. Estimated oil and gas development NO$_x$ emissions in the Southern Colorado Plateau Network as a percentage of total NO$_x$ emissions for the year 2008.
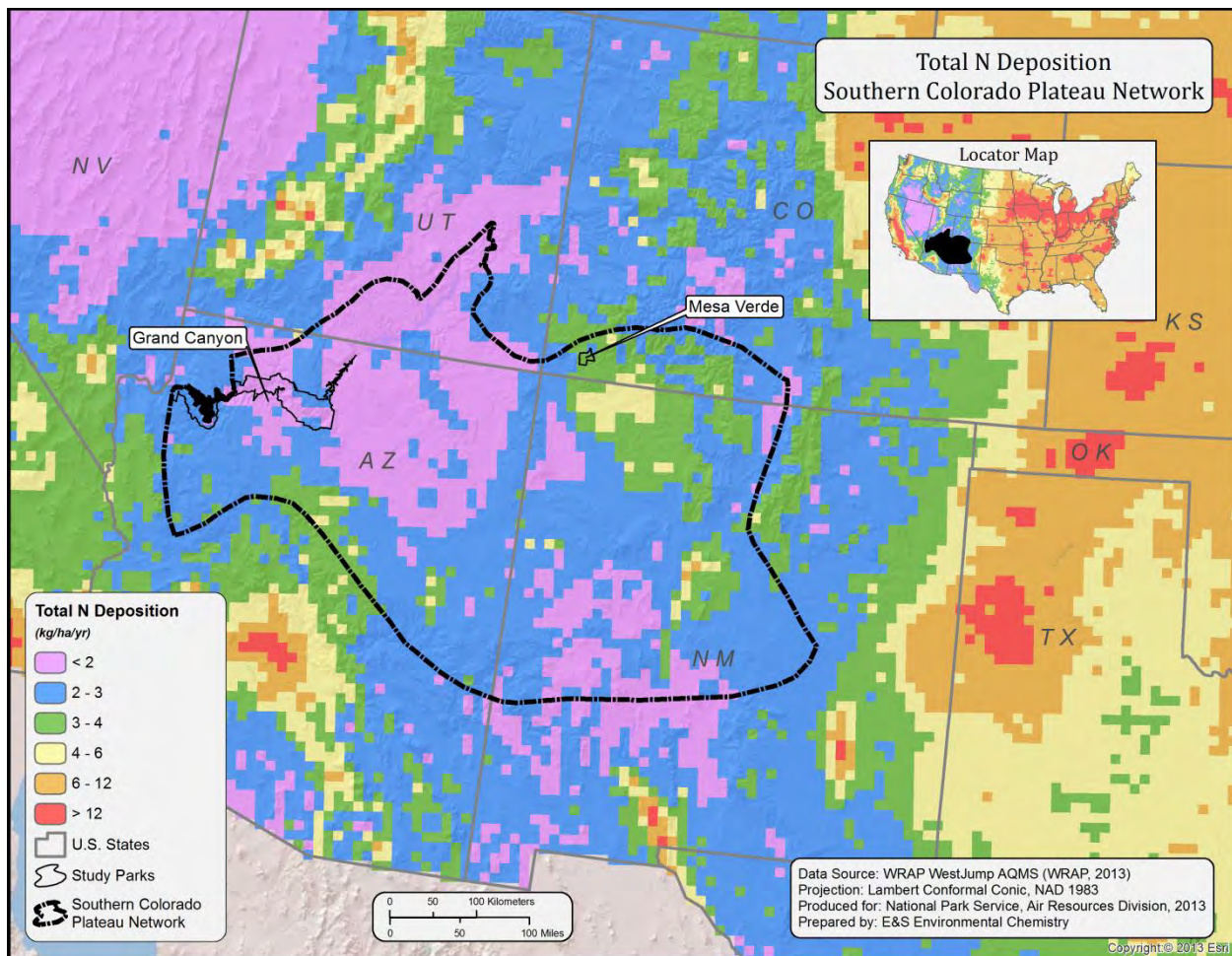
Map 9. CAMx model estimates of total wet plus dry N deposition (kg/ha/yr) for the year 2008 throughout the Southern Colorado Plateau Network.

Exceedance

Most of Mesa Verde NP receives ambient N deposition that is in exceedance of the lower estimate of the nutrient-N empirical CL reported by Allen and Geiser (2011); the remainder is largely below, but within 1 kg N/ha/yr of, exceedance (Map 10). Although none of Grand Canyon NP is in exceedance, most of the western third of the park and scattered portions of the remainder of the park are close (within 1 kg N/ha/yr) to exceedance (Map 11). Thus, it is likely that nutrient effects caused by N deposition may occur under ambient N loading at Mesa Verde NP, and significant portions of Grand Canyon NP may be at risk if N emissions from electricity generating units (EGUs), oil and gas exploration and production, or other sources, increase appreciably over 2008 levels.
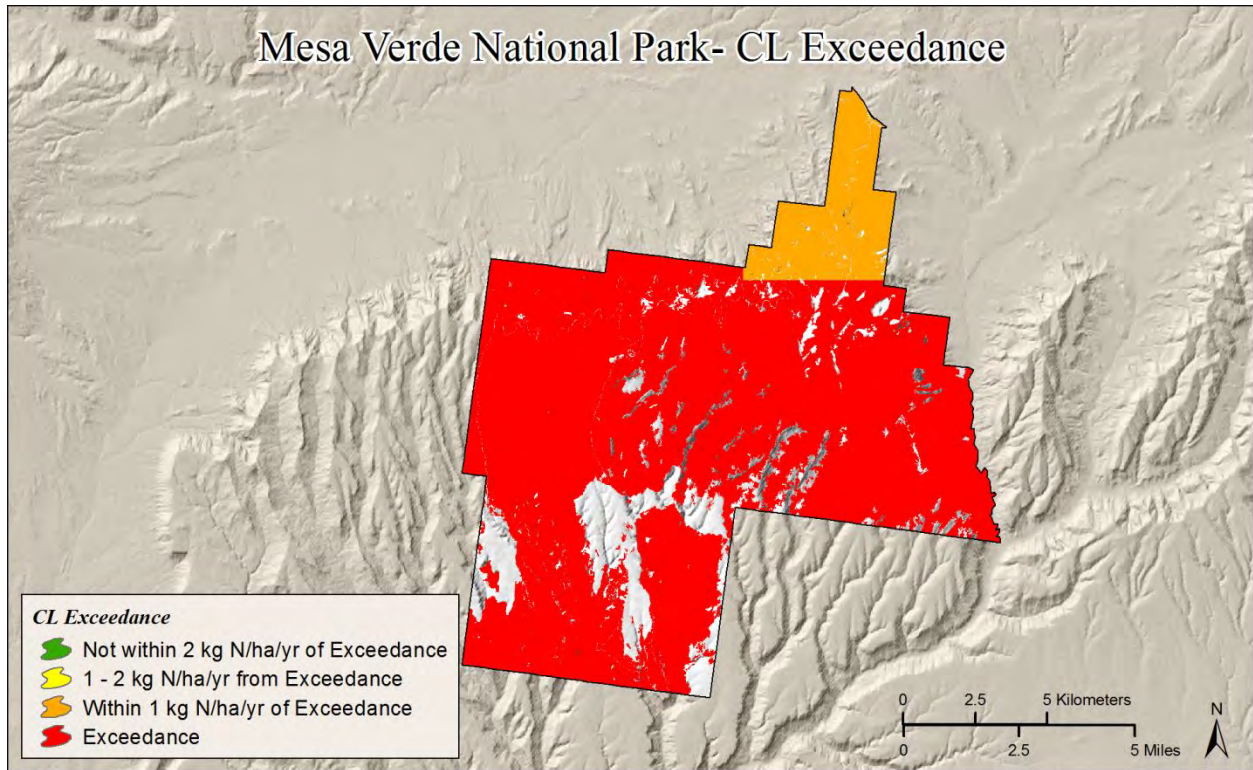
Map 10. Locations of vegetation types in Mesa Verde National Park that exhibit CL exceedance due to nutrient-N enrichment. The area of sensitive vegetation is labeled red if total N deposition in year 2008 is in exceedance of the lower limit of the critical load (CL) estimated by Allen and Geiser (2011). The remaining areas of sensitive vegetation are classified according to the extent to which total N deposition is below the CL. Areas showing the background color do not contain vegetation types identified by Allen and Geiser (2011) as being highly sensitive to nutrient-N enrichment.
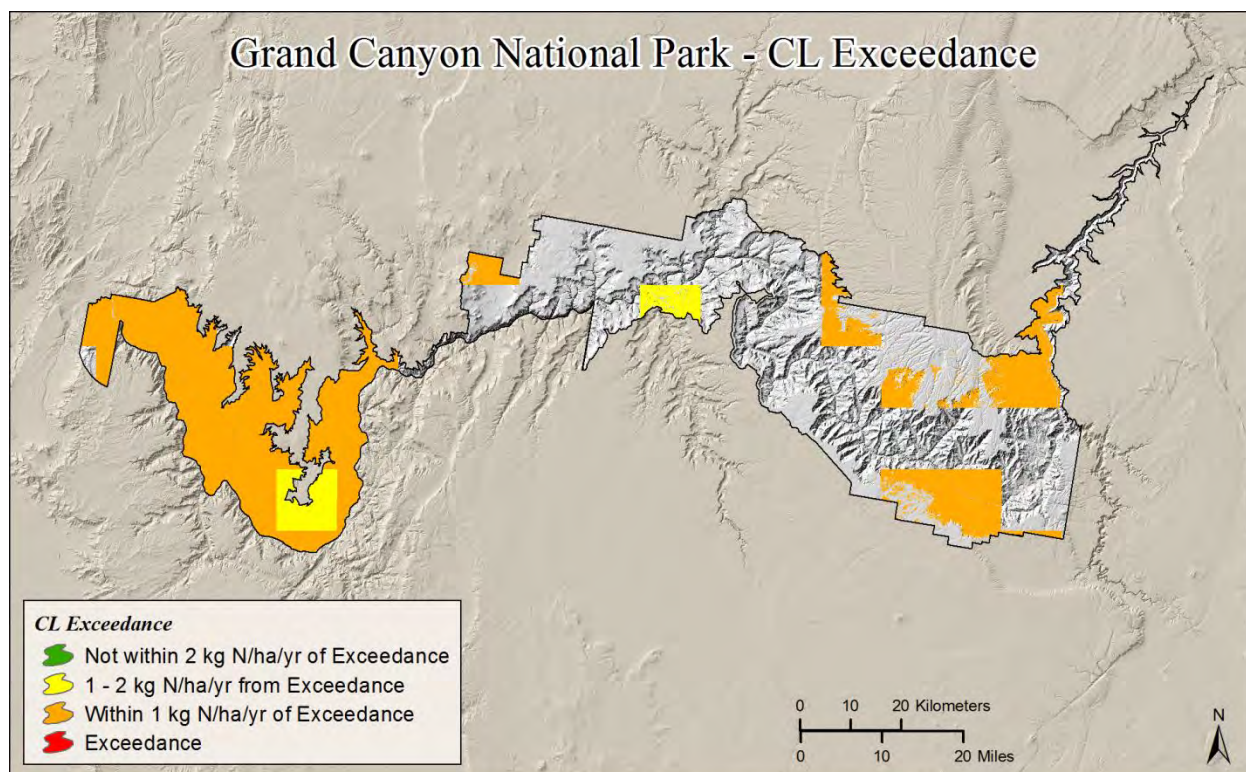
Map 11. Locations of vegetation types in Grand Canyon National Park that exhibit CL exceedance due to nutrient-N enrichment. The area of sensitive vegetation is labeled red if total N deposition in year 2008 is in exceedance of the lower limit of the critical load (CL) estimated by Allen and Geiser (2011). The remaining areas of sensitive vegetation are classified according to the extent to which total N deposition is below the CL. Areas showing the background color do not contain vegetation types identified by Allen and Geiser (2011) as being highly sensitive to nutrient-N enrichment.

Northern Colorado Plateau Network

NCPN contains five national park lands that are subjects of this analysis: Arches NP, Black Canyon of the Gunnison NP, Canyonlands NP, Colorado NM, and Dinosaur NM. Empirical critical loads are summarized below for each.

Dose-Response and Critical Loads

Terrestrial resources in the parks in NCPN are expected to be highly sensitive to nutrient-N enrichment from atmospheric N deposition, given the widespread distribution of arid and semi-arid plant communities. Because N deposition levels are variable in this network and may be increasing in response to oil and gas exploration and production, agriculture, or other sources, impacts of excess N may occur now or in the future at some locations.

Only limited research has been conducted on the effects of atmospheric N deposition on nutrient dynamics of native plant communities in the NCPN. Some research has been conducted on lichens and microbiotic soil crusts (Belnap 1991, Belnap et al. 2008, McCrackin et al. 2008). A fertilization experiment in an arid grassland demonstrated a large difference in the response of native and non-native plants to added N. For two years, plots were treated with 0, 10, 20, or 40 kg N/ha/yr as a

potassium nitrate (KNO$_3$) solution (Schwinning et al. 2005). Galleta (Hilaria jamesii) and Indian ricegrass (Oryzopis hymenoides) showed no increase in leaf photosynthesis or tiller size, but ricegrass showed a 50% increase in tiller density in the second year at the 20 and 40 kg N/ha/yr application levels. For both species, the increased N application hastened the onset of water stress. Unexpectedly, non-native Russian thistle (Salsola iberica) showed rapid growth response to the highest fertilization rate in the first summer when rainfall was above average. Schwinning and colleagues (2005) suggested that the timing and amount of N deposition could facilitate noxious weed invasion and thus change community composition in arid grasslands. Nevertheless, interactions with water availability may also be important.

Portions of Arches NP, Black Canyon of the Gunnison NP, Canyonlands NP, Colorado NM, and Dinosaur NM that are covered by woodland, shrubland, and grassland are shown in Maps 12–16. The relative abundance of these plant community types varies by park. Each was estimated by Allen and Geiser (2011) to be sensitive to nutrient-N deposition and to have CL in the range of 3 to 8.4 kg N/ha/yr. Locations in the park where these plant communities occur and where ambient N deposition is higher than 3 kg N/ha/yr are in exceedance of the CL for protecting these potentially N-sensitive plant communities against change in community composition. This same CL (3 kg N/ha/yr) was also judged by Allen and Geiser (2011) to be applicable to protection of lichen community composition in desert plant communities such as are found in these NCPN parks.
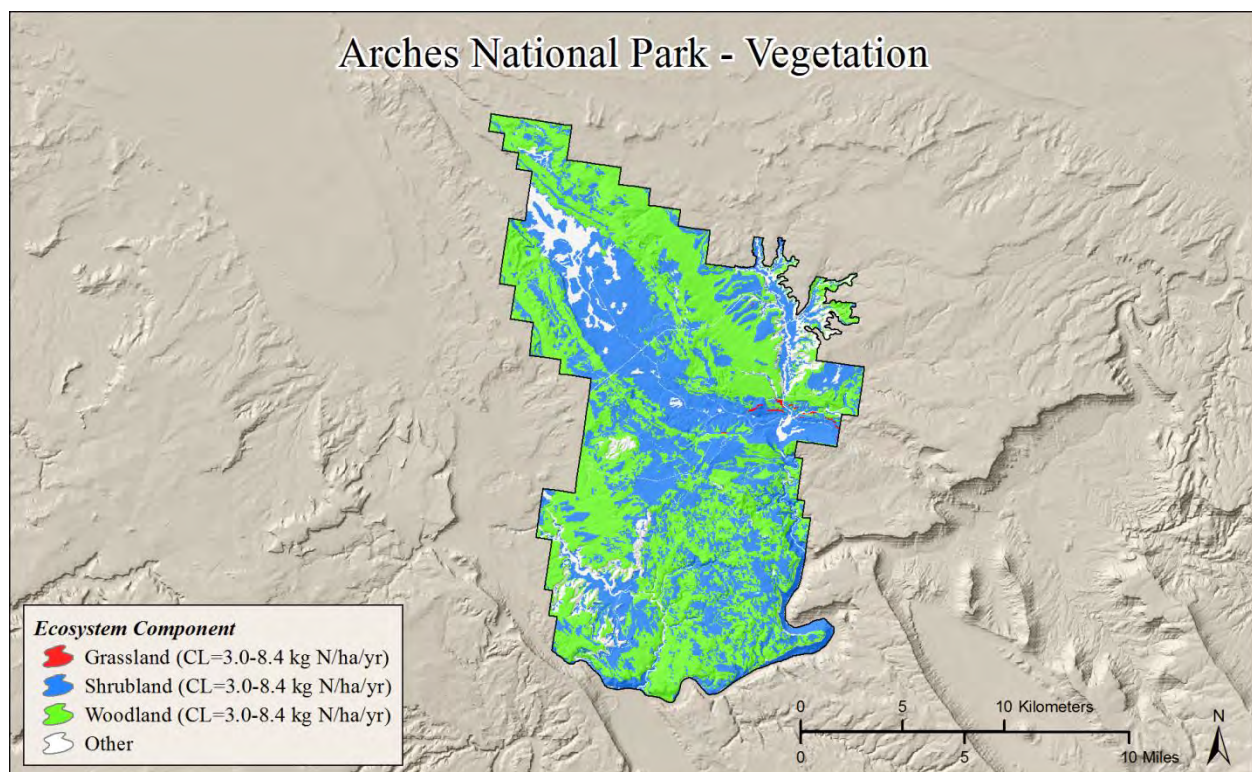
Map 12. Locations in Arches National Park having vegetation types characterized by the presence of plant communities expected to be most sensitive to nutrient-N deposition, based on nutrient critical load values compiled by Allen and Geiser (2011).
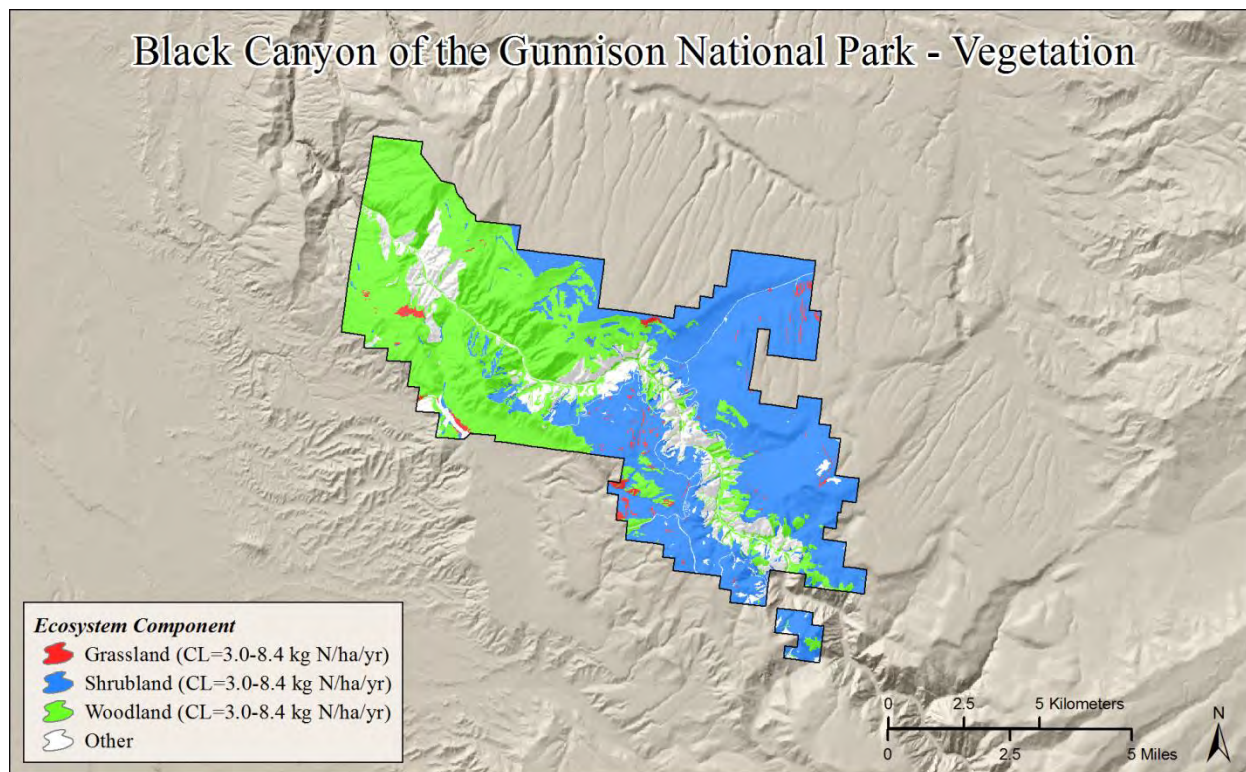
Map 13. Locations in Black Canyon of the Gunnison National Park having vegetation types characterized by the presence of plant communities expected to be most sensitive to nutrient-N deposition, based on nutrient critical load values compiled by Allen and Geiser (2011).
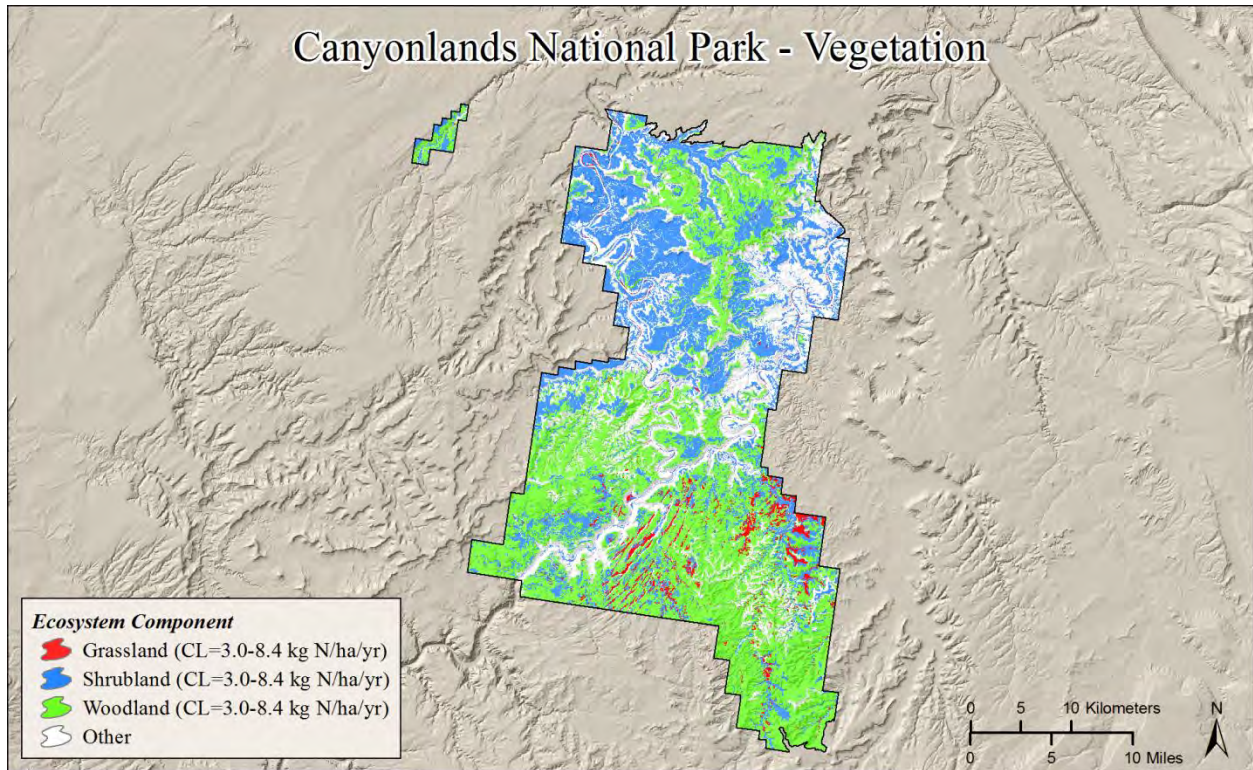
Map 14 Locations in Canyonlands National Park having vegetation types characterized by the presence of plant communities expected to be most sensitive to nutrient-N deposition, based on nutrient critical load values compiled by Allen and Geiser (2011).
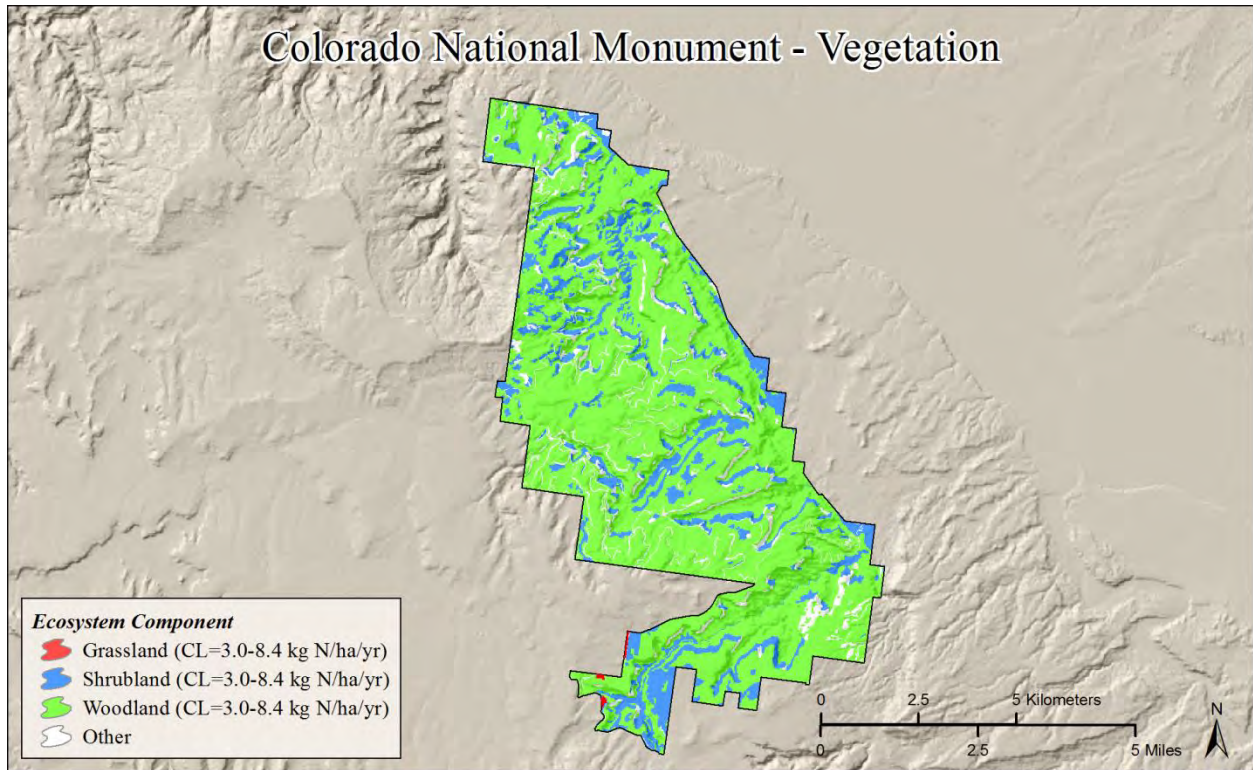
Map 15. Locations in Colorado National Monument having vegetation types characterized by the presence of plant communities expected to be most sensitive to nutrient-N deposition, based on nutrient critical load values compiled by Allen and Geiser (2011).
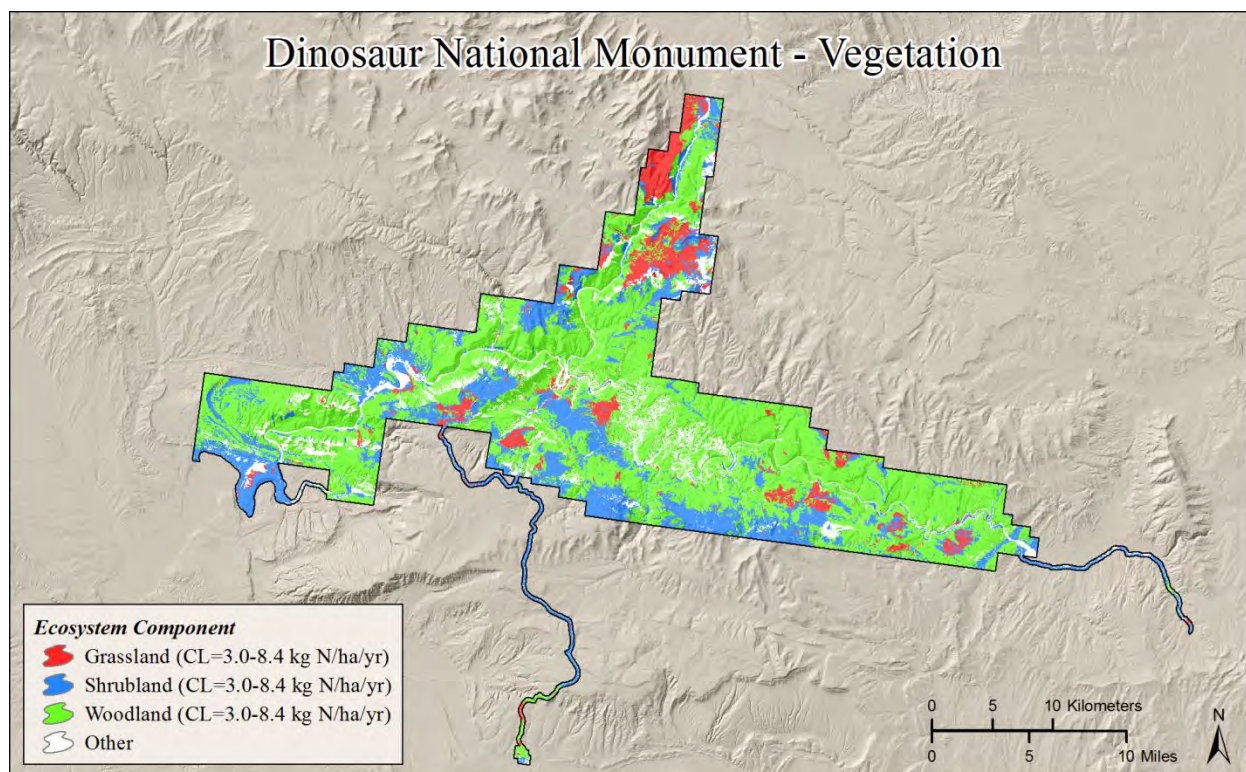
Map 16. Locations in Dinosaur National Monument having vegetation types characterized by the presence of plant communities expected to be most sensitive to nutrient-N deposition, based on nutrient critical load values compiled by Allen and Geiser (2011).

Emissions and Deposition

National Park units in the eastern half of NCPN are in close proximity to elevated NO_x emissions from oil and gas exploration and production, above 5,000 tpy in many counties based on estimates for 2008 (Map 17). One county located between Dinosaur NM and Colorado NM had between 10,000 and 24,000 tpy of oil and gas NO_x emissions. Of the parks in this network considered here, Dinosaur NM appears to be at greatest risk of receiving increased N deposition from oil and gas development. Many of the counties around Dinosaur NM emitted more than 1 ton/mi2/yr of NO_x from oil and gas development (Map 18). Total NO_x emissions from all source types is relatively high (> 10,000 tpy) at scattered locations throughout the network, especially near Dinosaur NM, Arches NP, and Canyonlands NP (Map 19). Based on 2008 data compiled by WestJump, oil and gas exploration and production accounts for an appreciable percentage (> 20%) of total NO_x emissions from several counties in NCPN, in proximity to all the park units considered in this analysis (Map 20). In some cases, the oil and gas contribution exceeds 60%. At other locations, relatively high total N deposition does not appear to be associated with known local NO_x emissions from oil and gas development, based on the data compiled for 2008 in the WestJump effort. Potentially important additional N sources include EGUs, agriculture, and mobile emissions.
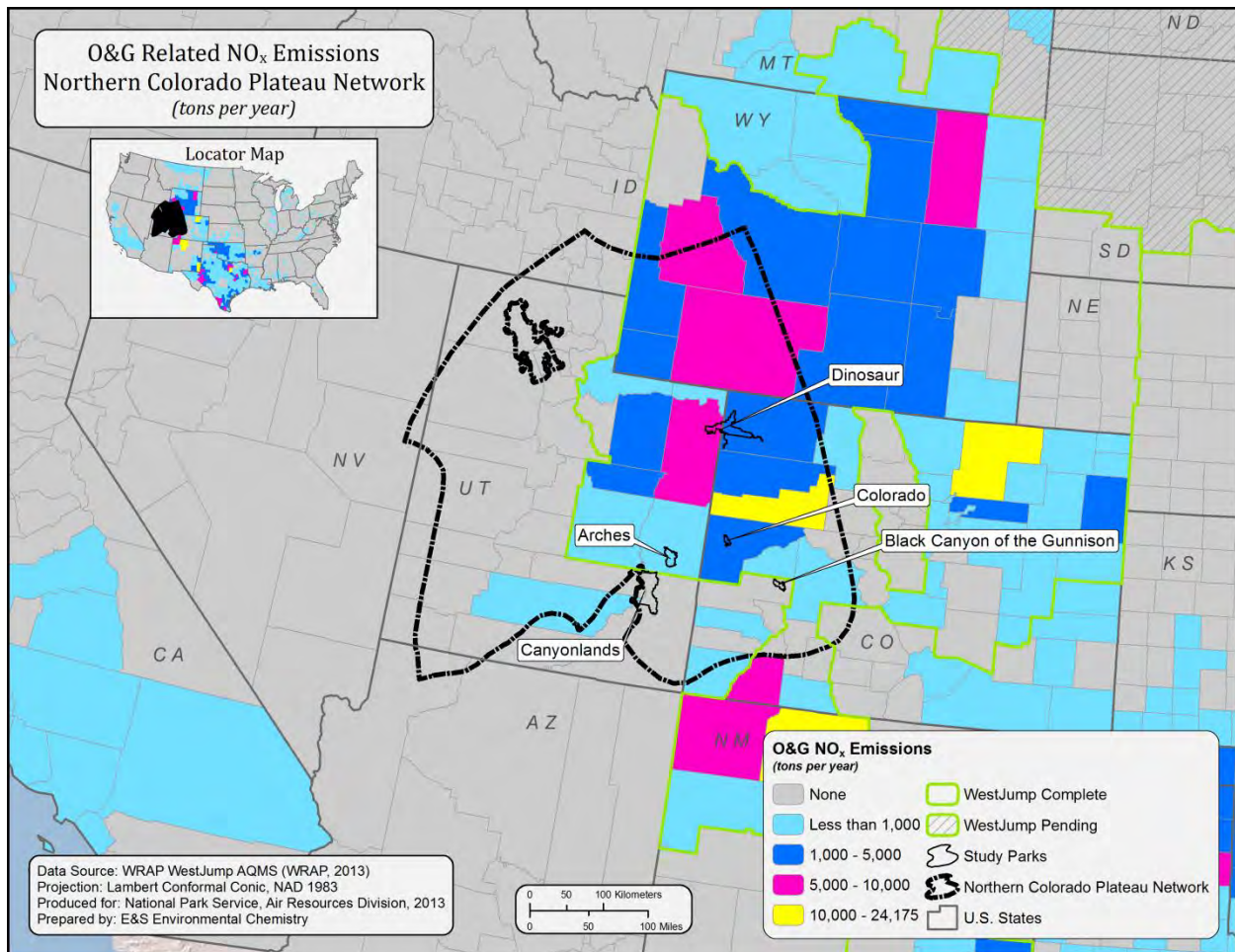
Map 17. Estimated oil and gas NO$_x$ emissions (tpy), by county in the Northern Colorado Plateau Network for the year 2008, which is the most recent year for which data are available at the time of this writing. Note that oil and gas emissions levels in some areas have likely increased in more recent years.
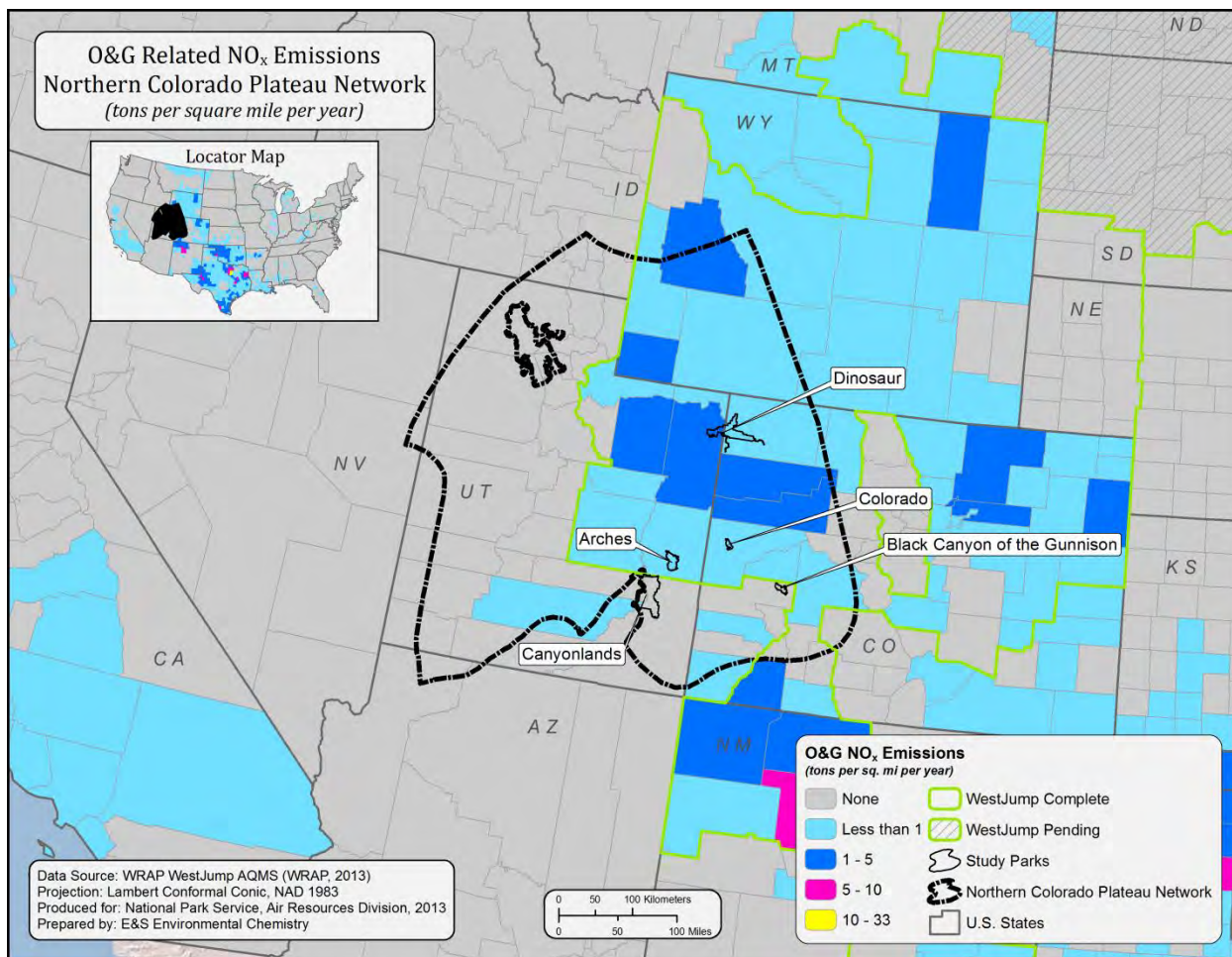
Map 18. Estimated oil and gas NO$_x$ emissions per unit land area (tons/mi^2/yr) in the Northern Colorado Plateau Network for the year 2008.
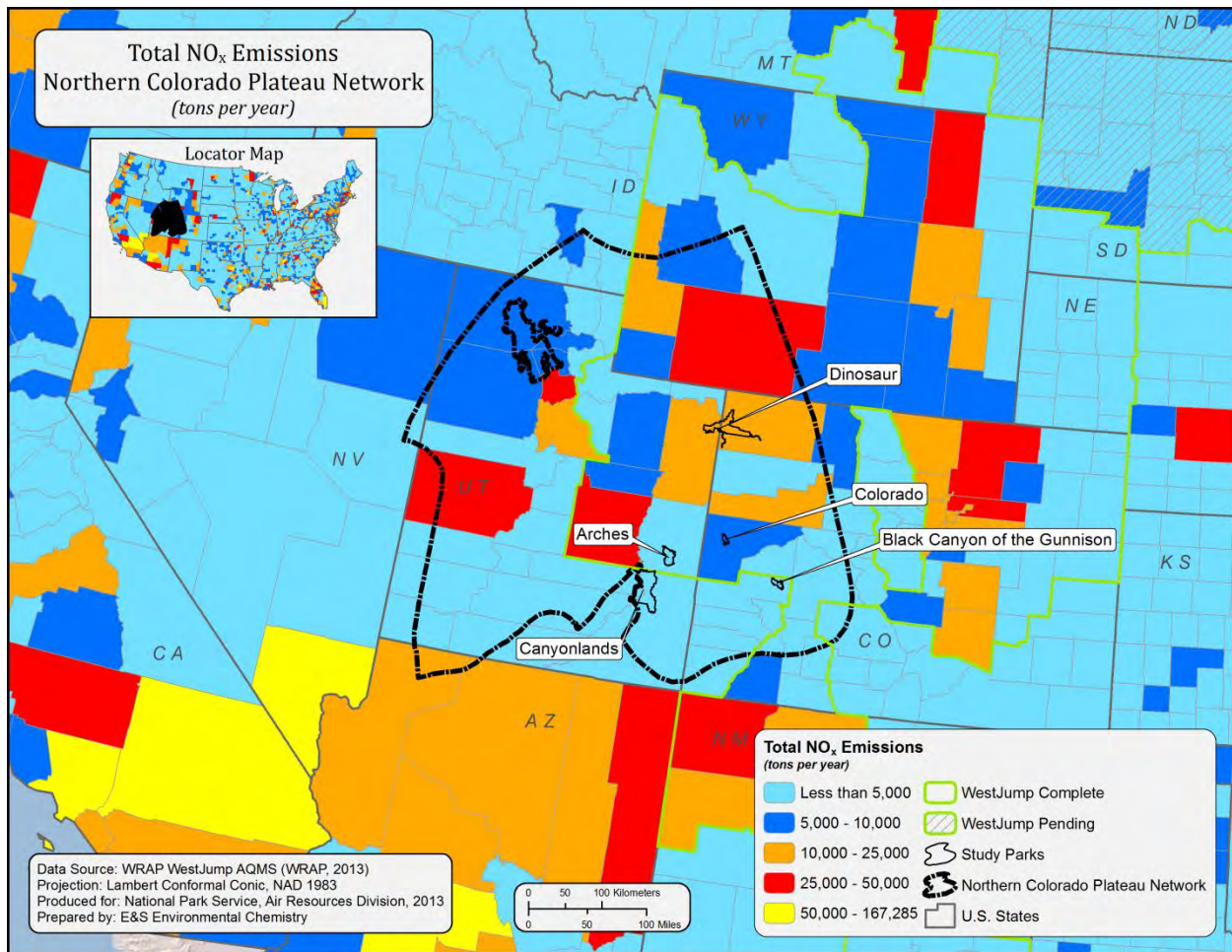
Map 19. Estimated NO$_x$ emissions from all sources (tpy), including oil and gas development, in the Northern Colorado Plateau Network for the year 2008.
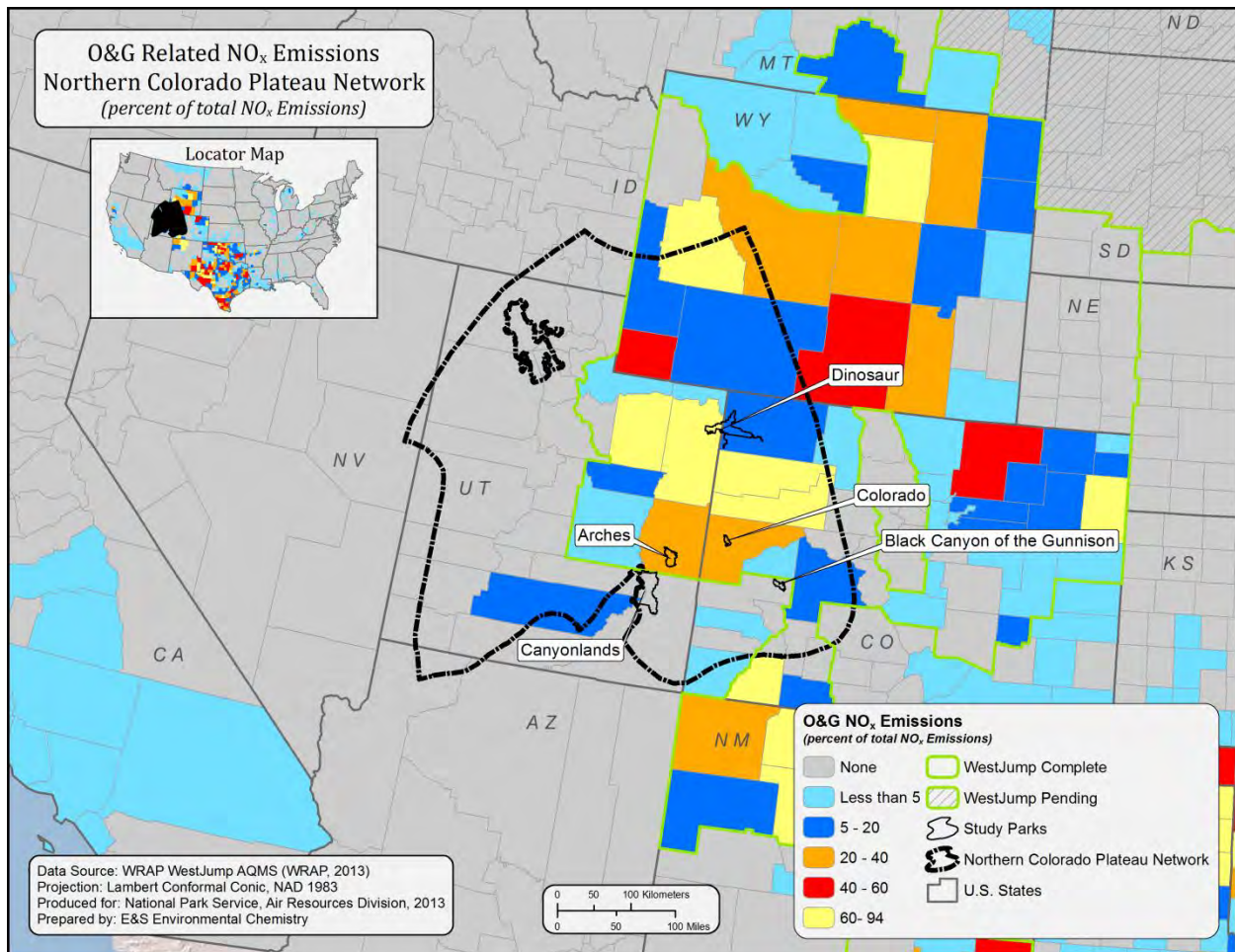
Map 20. Estimated oil and gas NO$_x$ emissions in the Northern Colorado Plateau Network as a percentage of total NO$_x$ emissions for the year 2008.

Map 21 shows CAMx model estimates of total wet plus dry N deposition for NGPN. Total N deposition within the network region is highly variable, ranging from less than 2 kg N/ha/yr around Canyonlands NP and Arches NP to more than 4 kg N/ha/yr in portions of northcentral Utah (Map 21).

Exceedence

Exceedance maps for the five national parks in NCPN that are subjects of study here are shown in Maps 22 through 26. There are no areas of nutrient-N exceedance in ARCH, DINO, or CANY, and only a small sliver of land in COLM that is in exceedance. In contrast, the northern half of BLCA is in exceedance and therefore at increased risk of ecological effects associated with N overfertilization. Essentially all of COLM and DINO and the southern half of both ARCH and BLCA receive ambient N deposition that is below, but close (within 1 kg N/ha/yr) to, exceedance. Most of CANY is within 2 kg N/ha/yr of exceedance. Thus, any appreciable increase in N emissions, from oil and gas exploration and production, agriculture, EGU, or other sources, may increase the risk of excess nutrient effects throughout most of the park lands within NCPN that are under study in this analysis.
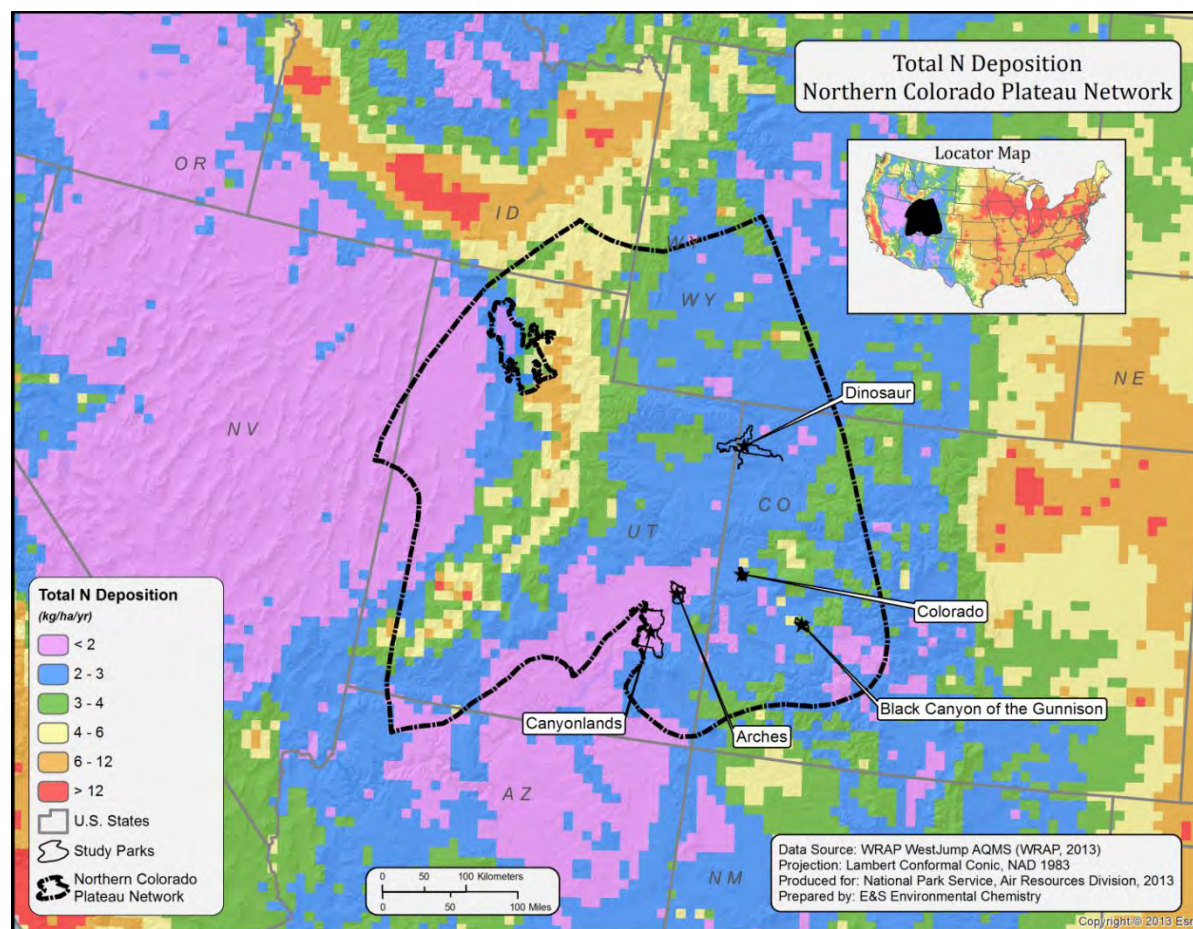
Map 21. CAMx model estimates of total wet plus dry N deposition (kg/ha/yr) for the year 2008 throughout the Northern Colorado Plateau Network.
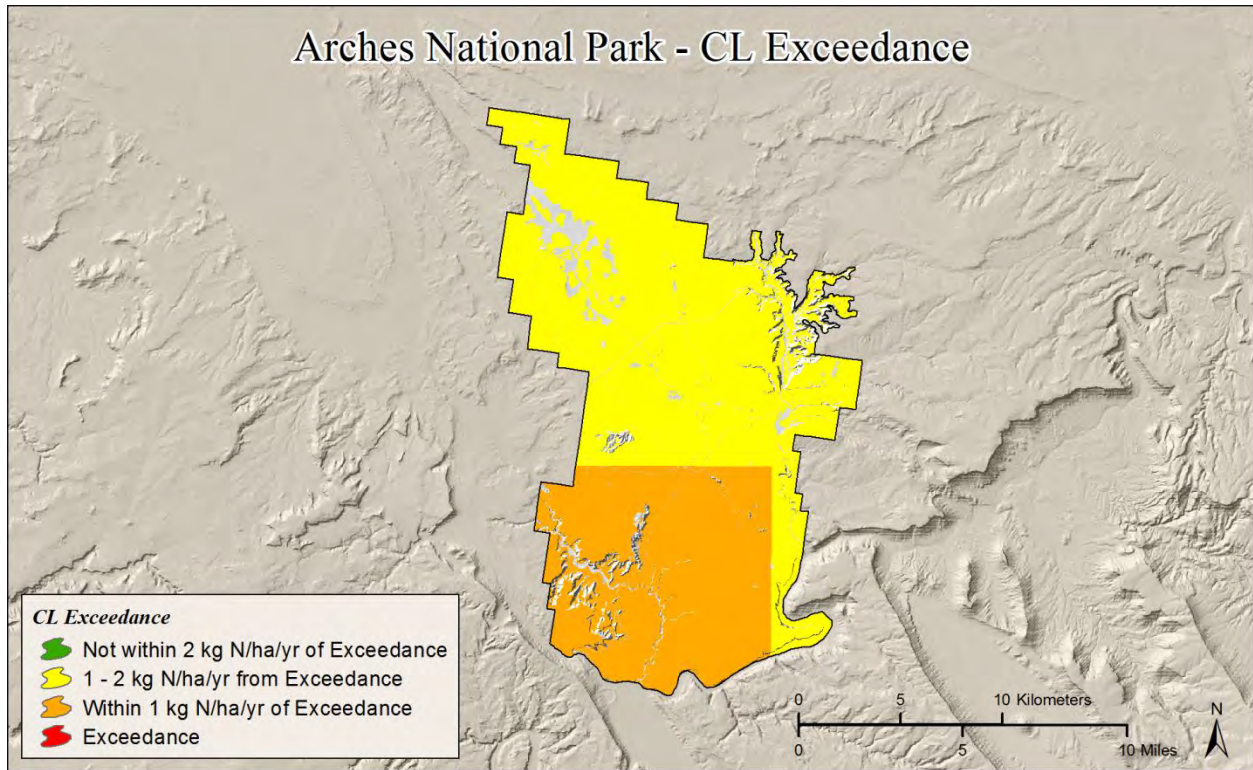
Map 22. Locations of vegetation types in Arches National Park that exhibit CL exceedance due to nutrient-N enrichment. The area of sensitive vegetation is labeled red if total N deposition in year 2008 is in exceedance of the lower limit of the critical load (CL) estimated by Allen and Geiser (2011). The remaining areas of sensitive vegetation are classified according to the extent to which total N deposition is below the CL. Areas showing the background color do not contain vegetation types identified by Allen and Geiser (2011) as being highly sensitive to nutrient-N enrichment.
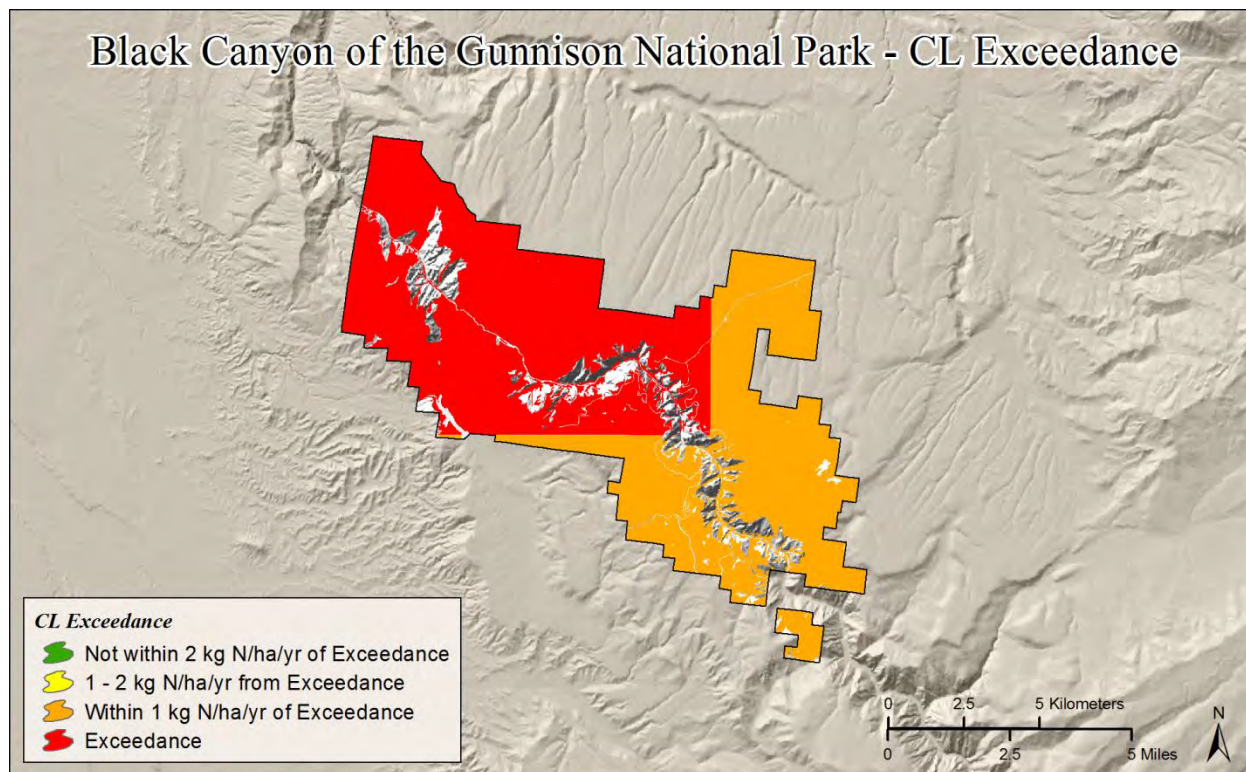
Map 23. Locations of vegetation types in Black Canyon of the Gunnison National Park that exhibit CL exceedance due to nutrient-N enrichment. The area of sensitive vegetation is labeled red if total N deposition in year 2008 is in exceedance of the lower limit of the critical load (CL) estimated by Allen and Geiser (2011). The remaining areas of sensitive vegetation are classified according to the extent to which total N deposition is below the CL. Areas showing the background color do not contain vegetation types identified by Allen and Geiser (2011) as being highly sensitive to nutrient-N enrichment.
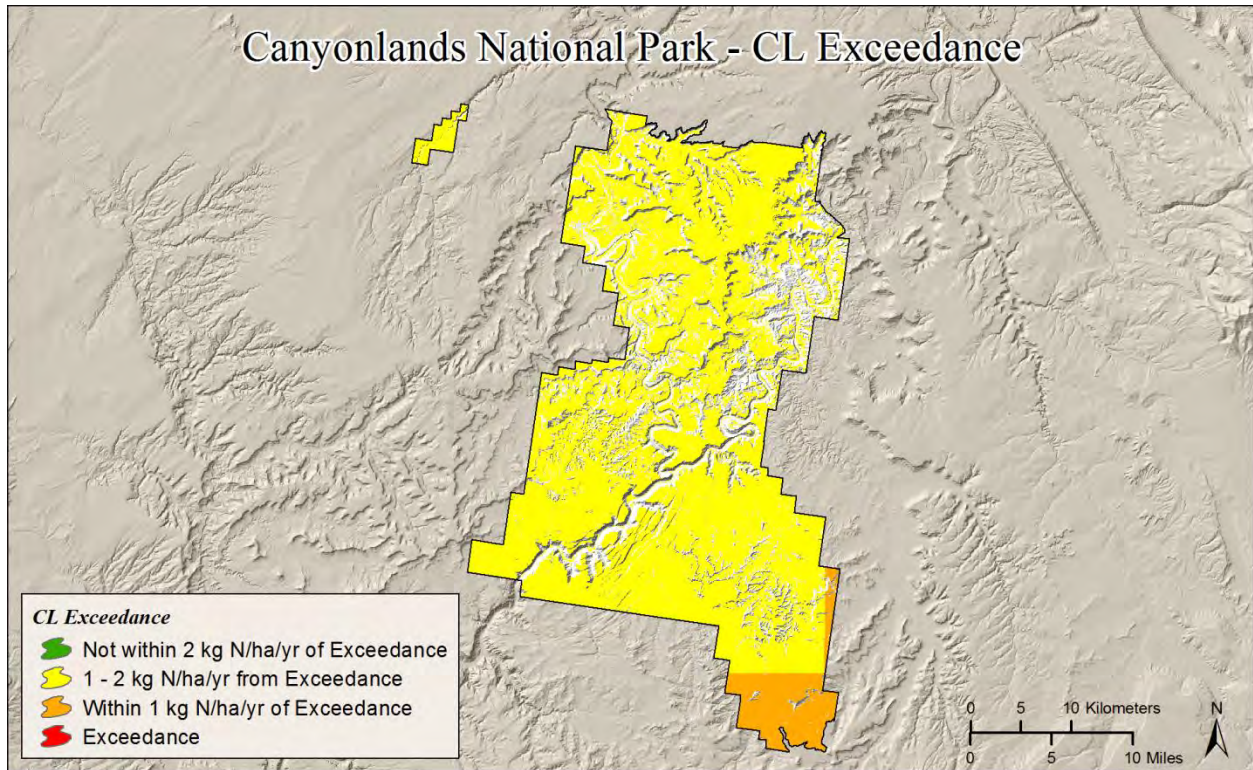
Map 24. Locations of vegetation types in Canyonlands National Park that exhibit CL exceedance due to nutrient-N enrichment. The area of sensitive vegetation is labeled red if total N deposition in year 2008 is in exceedance of the lower limit of the critical load (CL) estimated by Allen and Geiser (2011). The remaining areas of sensitive vegetation are classified according to the extent to which total N deposition is below the CL. Areas showing the background color do not contain vegetation types identified by Allen and Geiser (2011) as being highly sensitive to nutrient-N enrichment.
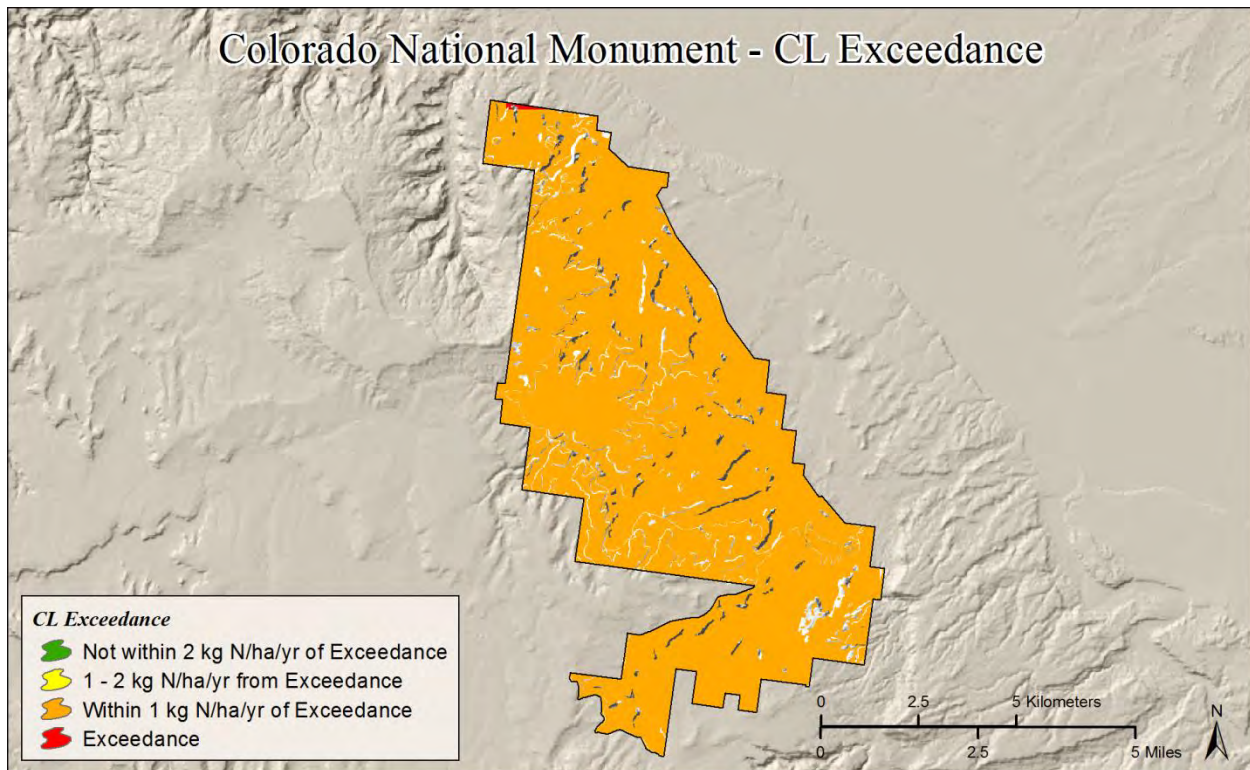
Map 25. Locations of vegetation types in Colorado National Monument that exhibit CL exceedance due to nutrient-N enrichment. The area of sensitive vegetation is labeled red if total N deposition in year 2008 is in exceedance of the lower limit of the critical load (CL) estimated by Allen and Geiser (2011). The remaining areas of sensitive vegetation are classified according to the extent to which total N deposition is below the CL. Areas showing the background color do not contain vegetation types identified by Allen and Geiser (2011) as being highly sensitive to nutrient-N enrichment.
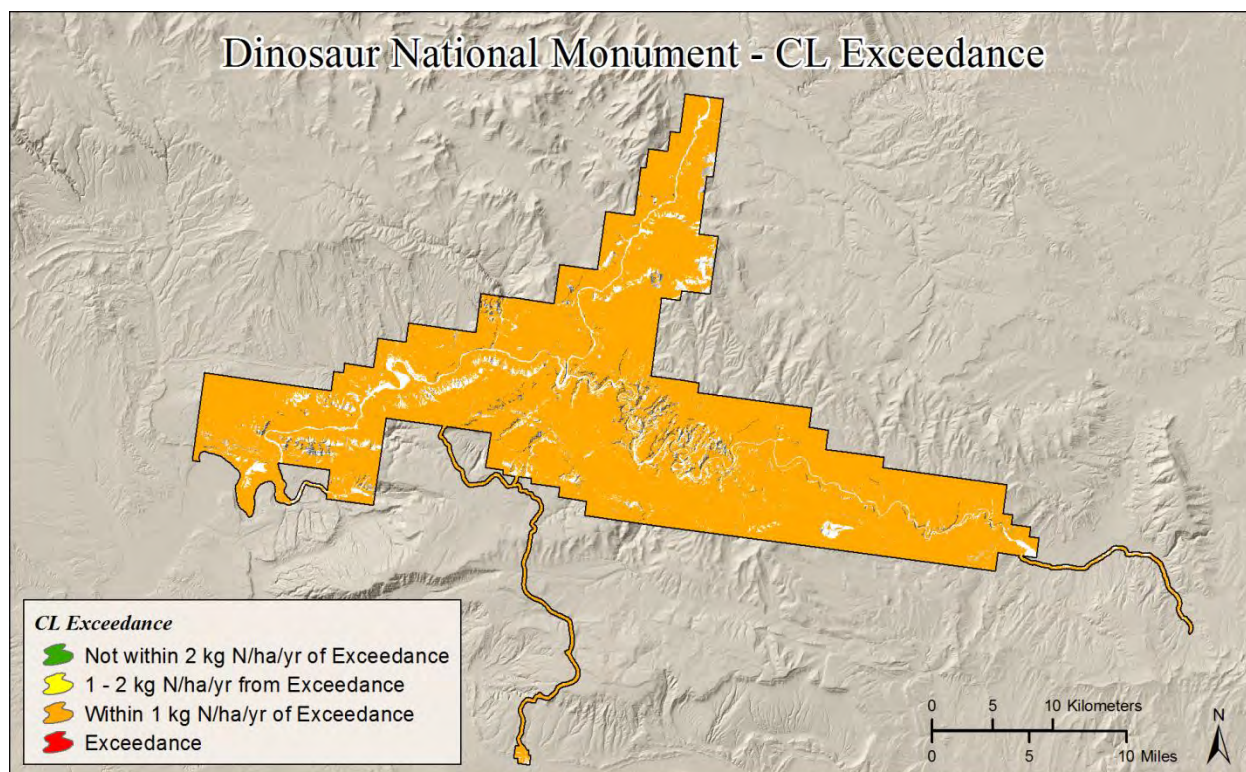
Map 26. Locations of vegetation types in Dinosaur National Monument that exhibit CL exceedance due to nutrient-N enrichment. The area of sensitive vegetation is labeled red if total N deposition in year 2008 is in exceedance of the lower limit of the critical load (CL) estimated by Allen and Geiser (2011). The remaining areas of sensitive vegetation are classified according to the extent to which total N deposition is below the CL. Areas showing the background color do not contain vegetation types identified by Allen and Geiser (2011) as being highly sensitive to nutrient-N enrichment.

Sonoran Desert Network
Dose Response and Critical Loads

In the driest desert ecosystems, such as within the Sonoran Desert Network, water is generally more limiting than N. Nevertheless, enhanced N could play a role in the recently observed invasion of some exotic plant species within parts of the Sonoran and nearby Mojave deserts. Brooks (2003) found that plant responses were influenced by specific rainfall events rather than by average annual rainfall, with the annual plants thriving in a year when high rainfall events triggered germination. In the Mojave Desert, creosote bush showed no increased growth response to experimental N additions (at 10 and 40 kg N/ha/yr as $Ca(NO_3)_2$, but did respond to increased water (Barker et al. 2006). Invasive annuals showed a greater response to elevated N than did native plant species.

The SODN contains one park (Saguaro NP) that was included in this analysis as a high-interest park. Map 27 shows CAMx model estimates of total wet plus dry N deposition for this network. Critical loads of atmospheric N deposition are analyzed below for Saguaro NP.
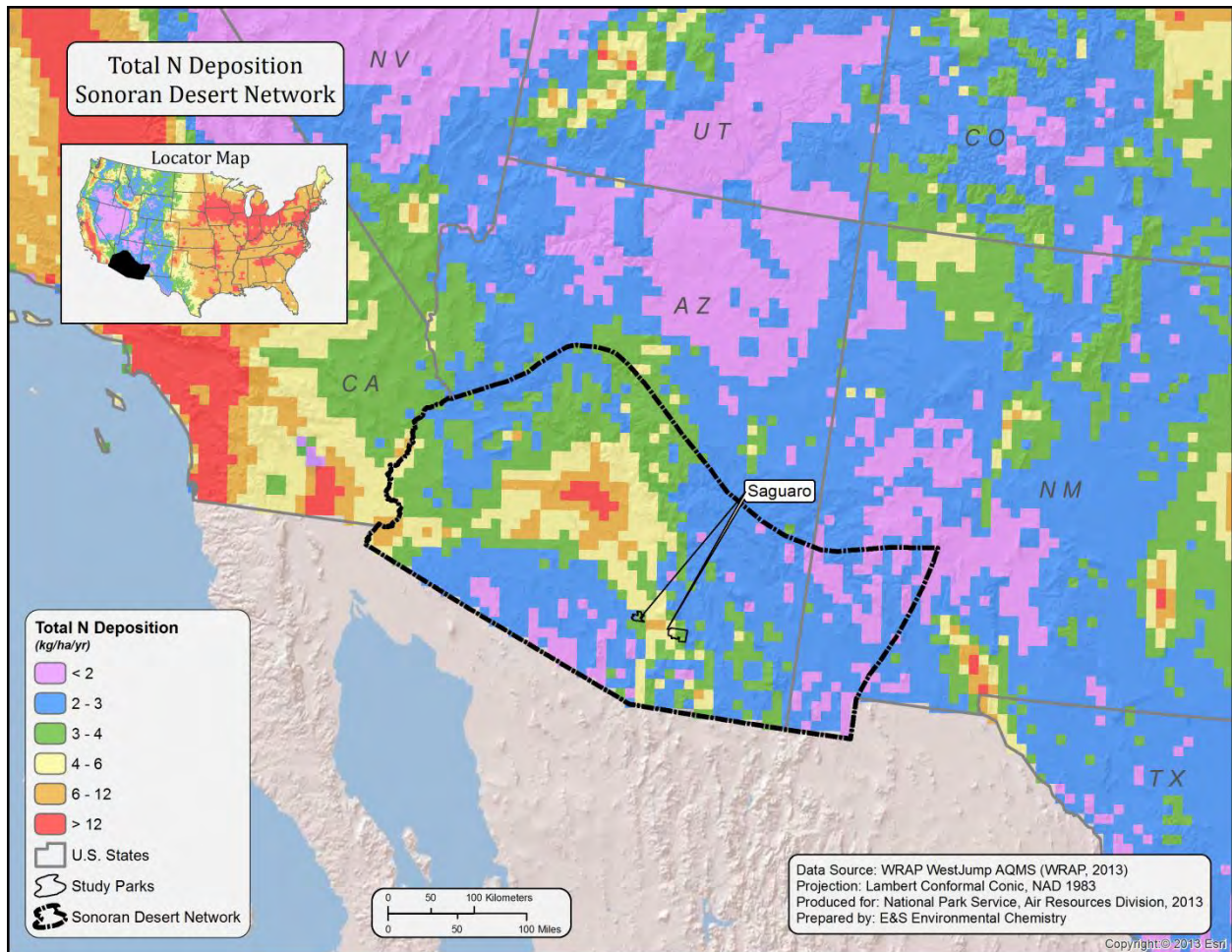
Map 27. CAMx model estimates of total wet plus dry N deposition (kg/ha/yr) for the year 2008 throughout the Sonoran Desert Network.

Saguaro National Park

Parts of Saguaro NP occur within the Sonoran Desert Level III ecoregion. In the lowland areas throughout the Sonoran Desert, N deposition would be expected to contribute to increased cover of non-native grasses and increased fire frequency and severity (Fenn and Allen 2011). Non-native grass invasion is an important management issue in Saguaro NP.

Desert lichens may be especially sensitive to N addition. Lichens near a large coal-fired plant in Page, Arizona showed chlorophyll degradation and reduced N fixation (Belnap 1991). In contrast, Marsh and Nash (1979) did not find effects on lichens near the Four Corners Power Plant on the Colorado Plateau.

Model simulations using the Patch Arid Land Simulator (PALS) suggested that plant biomass in the desert near Phoenix can increase with only a small increase in N supply (Shen et al. 2008). The issue

of increased plant biomass with increased N input is further confounded by the presence of nonnative invasive grass species, which are common in portions of the Sonoran Desert network.

Portions of Saguaro NP covered by woodland, shrubland, and grassland are shown in Map 28. Each of these plant community types was estimated by Allen and Geiser (2011) to be sensitive to nutrient-N deposition and to have CL in the range of 3 to 8.4 kg N/ha/yr. Locations in the park where these plant communities occur and where ambient N deposition is higher than 3 kg N/ha/yr are in exceedance of the CL for protecting these potentially N-sensitive plant communities against change in community composition. This same CL (3 kg N/ha/yr) was also judged by Allen and Geiser (2011) to be applicable to protection of lichen community composition in desert plant communities such as are found in Saguaro NP.

Emissions and Depositions

There are no WestJump emissions data available for the region around Saguaro NP or elsewhere in SODN. Based on emissions data included in the NEI, the only oil and gas development emissions within the network are found in the southeastern corner, and even there the emissions levels are estimated to be less than 1,000 tpy and in just one county (Map 29). Emissions per unit area are similarly low (Map 30). Although total NO_x emissions from all sources are moderately high in the counties near Saguaro NP, in some cases between 25,000 and 167,000 tpy (Map 31), oil and gas exploration and production does not appear to account for an appreciable percentage of those emissions through 2008 (Map 32). It is not clear if inclusion of more of the small sources and more recent data might alter that preliminary conclusion.

Total wet plus dry N deposition in SODN was less than 3 kg N/ha/yr at most locations. However, estimated N deposition was much higher around Phoenix and other portions of south central Arizona, in some areas in excess of 12 kg N/ha/yr (Map 27).
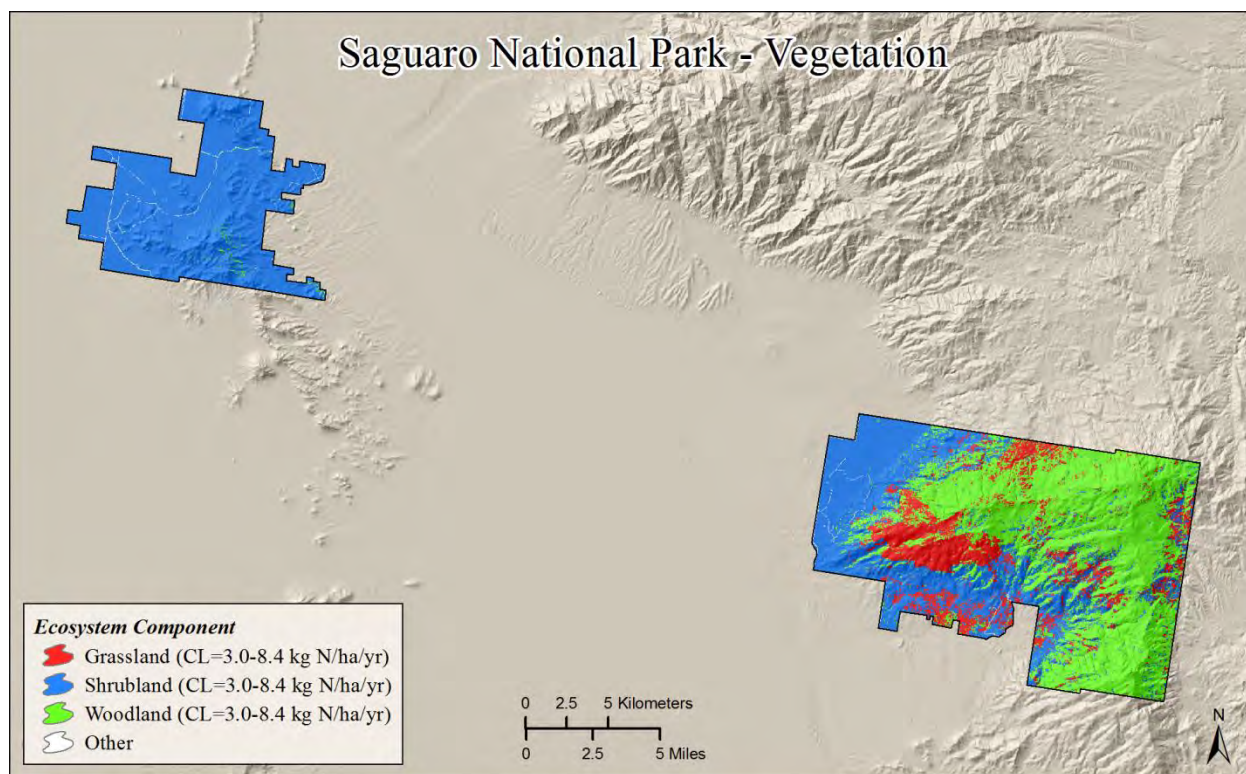
Map 28. Locations in Saguaro National Park having vegetation types characterized by the presence of plant communities expected to be most sensitive to nutrient-N deposition, based on nutrient critical load values compiled by Allen and Geiser (2011).
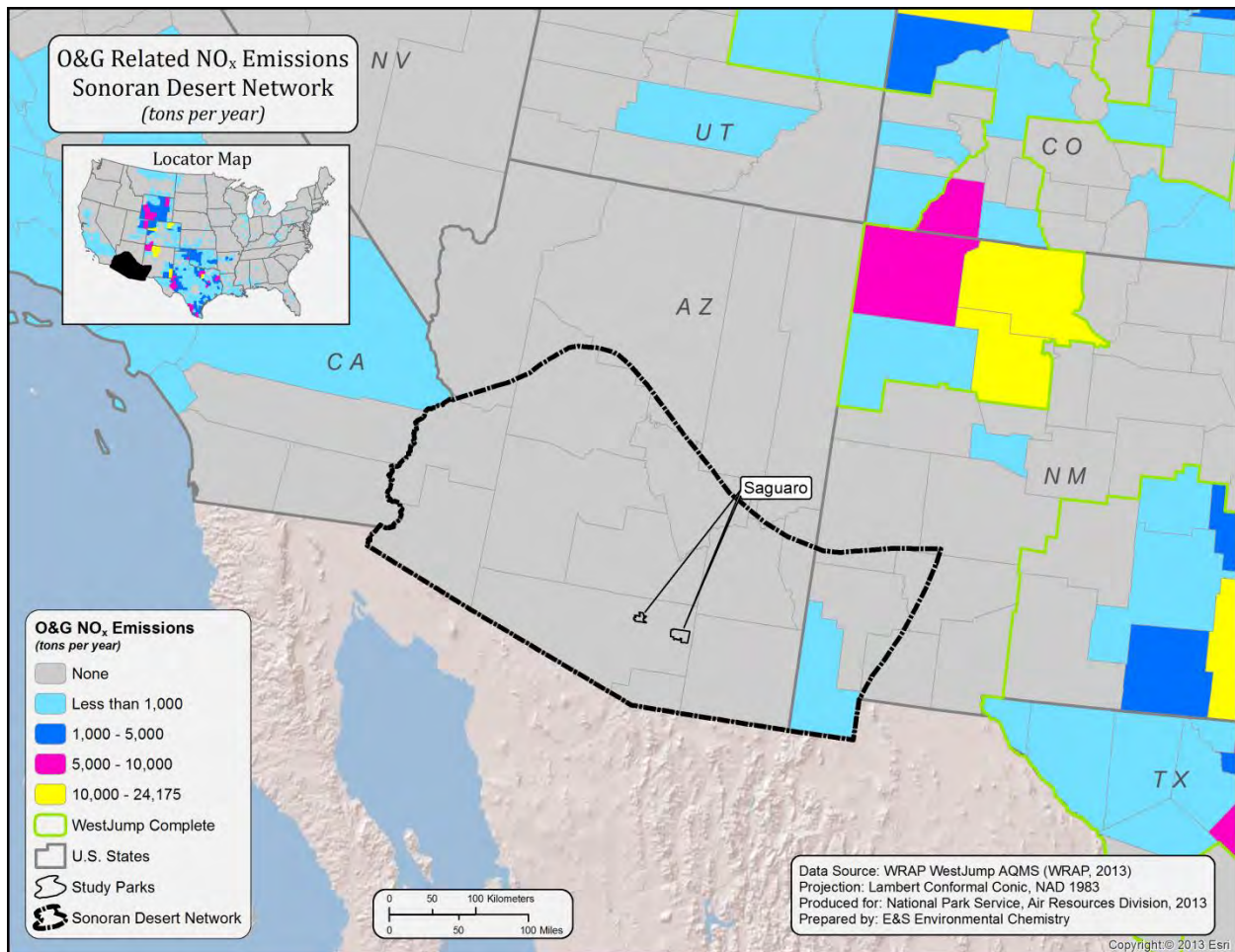
Map 29. Estimated oil and gas NO$_x$ emissions (tpy), by county, in the Sonoran Desert Network for the year 2008, which is the most recent year for which data are available. Note that oil and gas emissions levels in some areas have likely increased in more recent years.
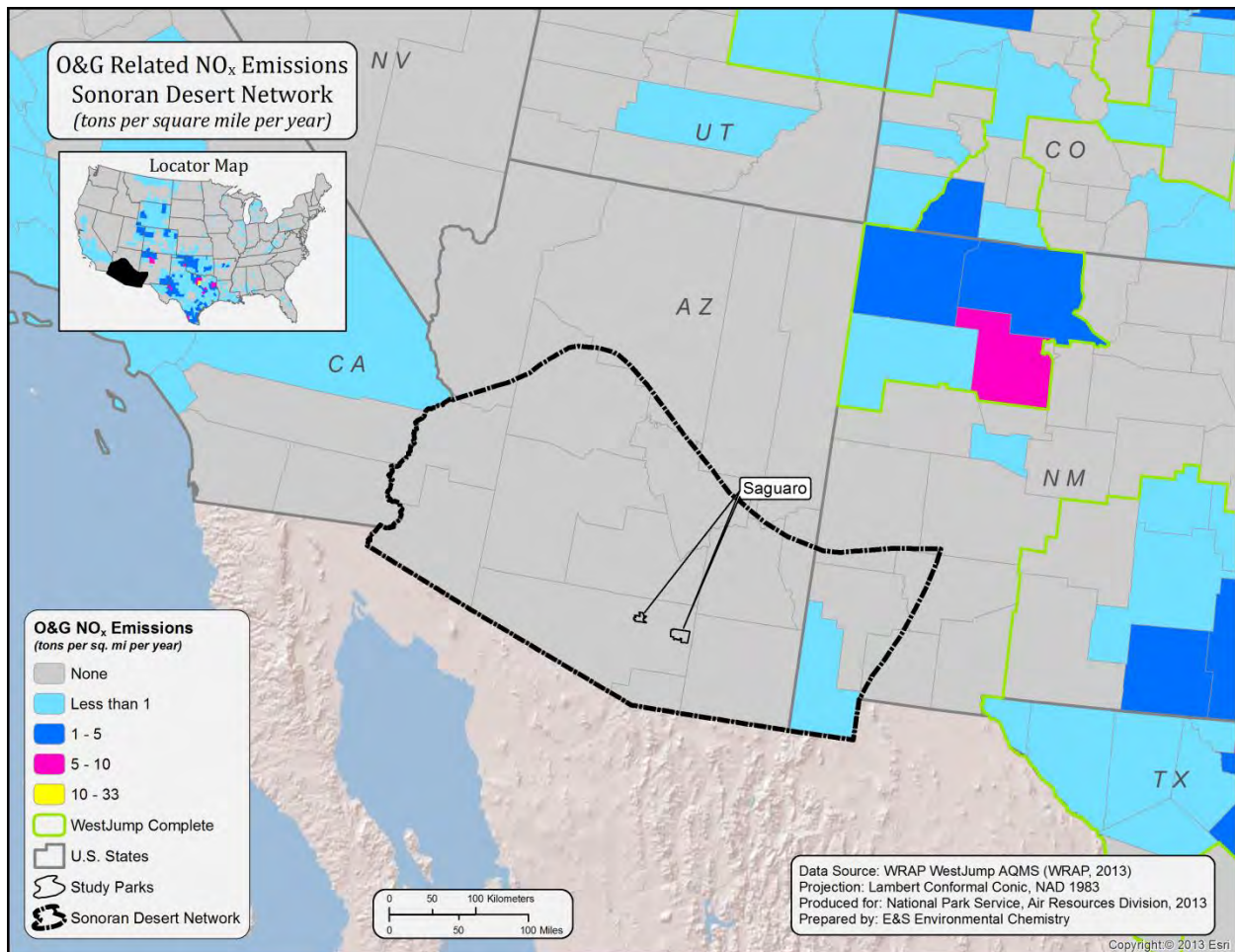
Map 30. Estimated oil and gas NO$_x$ emissions per unit land area (tons/mi^2/yr) in the Sonoran Desert Network for the year 2008.
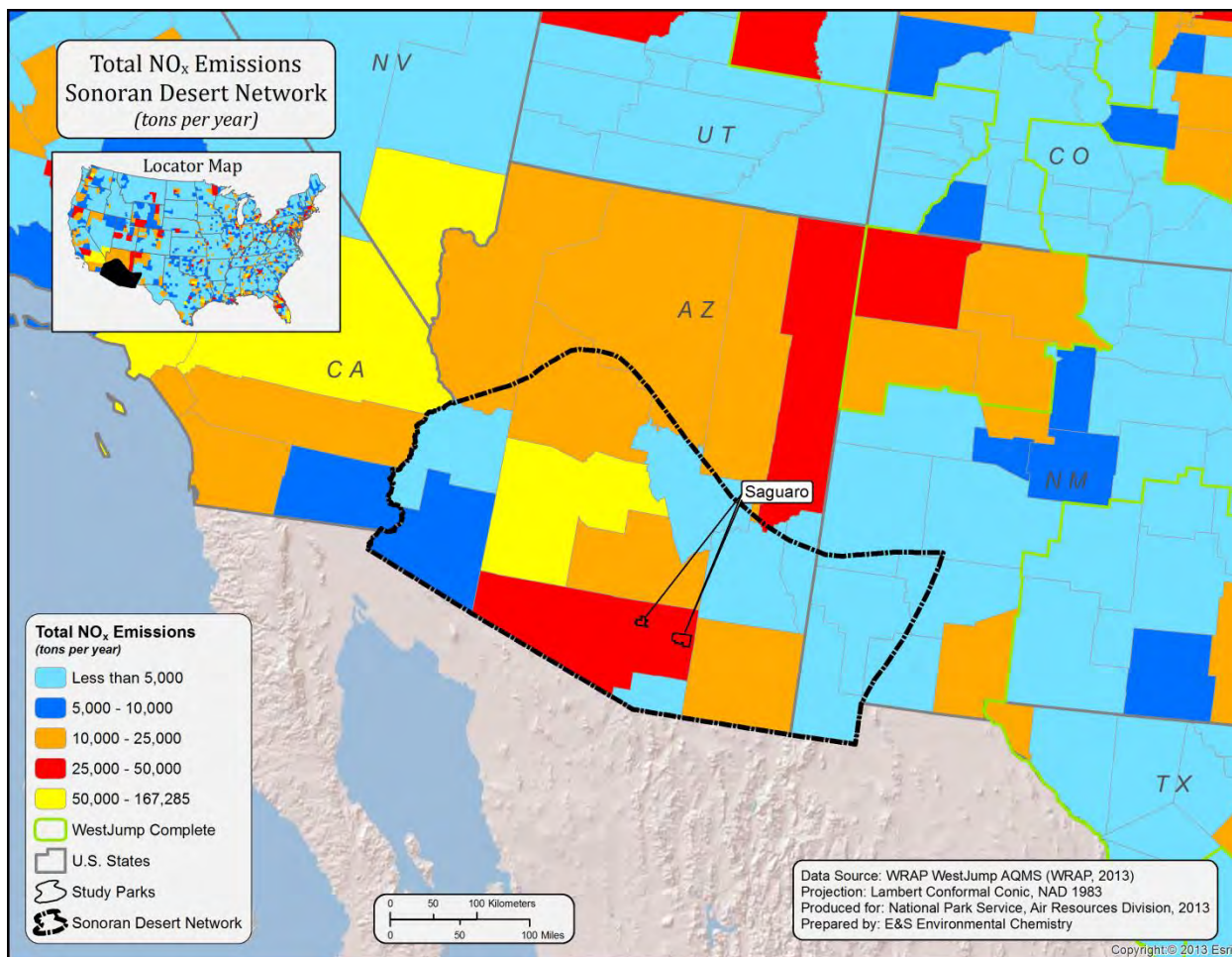
Map 31. Estimated NO$_x$ emissions from all sources (tpy), including oil and gas development, in the Sonoran Desert Network for the year 2008.
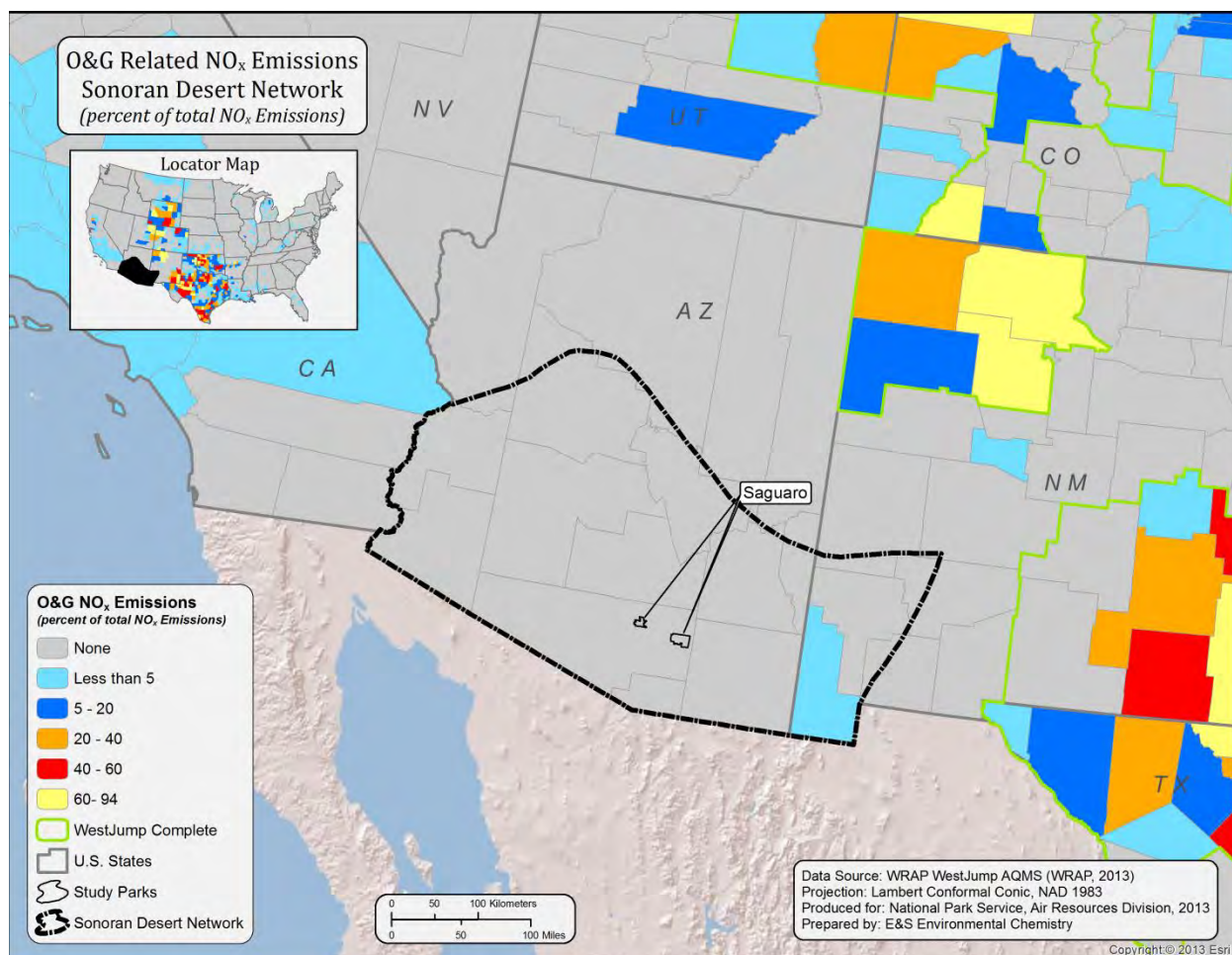
Map 32. Estimated oil and gas NO$_x$ emissions in the Sonoran Desert Network as a percentage of total NO$_x$ emissions for the year 2008.
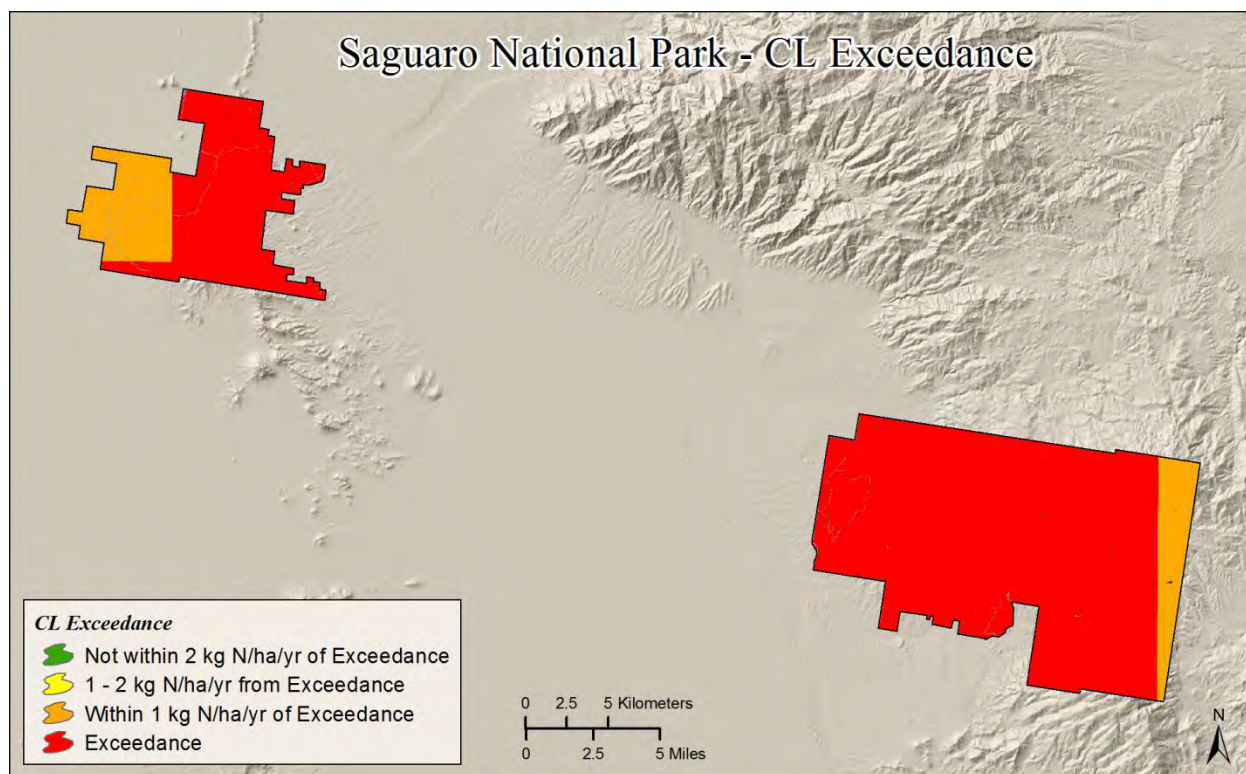
Map 33. Locations of vegetation types in Saguaro National Park considered sensitive to nutrient-N enrichment. The area of sensitive vegetation is labeled red if total N deposition in year 2008 is in exceedance of the lower limit of the critical load (CL) estimated by Allen and Geiser (2011). The remaining areas of sensitive vegetation are classified according to the extent to which total N deposition is below the CL. Orange indicates areas that are within 1 kg N/ha/yr of exceedance. Yellow indicates areas that are between 1 and 2 kg N/ha/yr from exceedance. Green represents areas that are not within 2 kg N/ha/yr of exceedance. Areas showing the background color do not contain vegetation types identified by Allen and Geiser (2011) as being highly sensitive to nutrient-N enrichment.

Exceedance

The majority of the parkland in Saguaro NP, in both the East and West units, received ambient N deposition that is in exceedance of the nutrient-N CL (Map 33). The remainder of the park received N deposition below, but within 1 kg N/ha/yr of, the CL. Thus, there is a reasonable likelihood of adverse impacts at Saguaro NP under ambient N deposition loading and an increased likelihood that further increases in N emissions in and near Saguaro NP could cause ecological changes related to excess nutrient deposition.

Northern Forests Ecoregion

Based on recent research by Thomas et al. (2010), northern hardwood forests that predominate in the northern portions of Minnesota, Wisconsin, and Michigan are expected to show growth enhancement by some, but not all, tree species in response to ongoing or future atmospheric N addition. For example, eastern hardwood forests from Wisconsin to Maine and south to Virginia exhibited increased growth with increasing N deposition across a depositional gradient from about 3 to 11 kg N/ha/yr (Thomas et al. 2010). Nevertheless, responses were species-specific. Growth increases were

most pronounced for red maple (*Acer rubrum*), sugar maple (*Acer saccharum*), white ash (*Fraxinus americana*), yellow poplar (*Liriodendron tulipifera*), black cherry (*Prunus serotina*), balsam fir (*Abies balsamea*), pignut hickory (*Carya glabra*), eastern white pine (*Pinus strobus*), quaking aspen (*Populus tremuloides*), northern red oak (*Quercus rubra*), and scarlet oak (*Quercus coccinea*). Other tree species showed negative response to N addition. Growth decreased by a statistically significant amount with increasing deposition for red pine (*Pinus resinosa*), red spruce (*Picea rubens*), and northern white cedar (*Thuja occidentalis*). Mortality increased with increasing deposition for yellow birch (*Betula alleghaniensis*), eastern white pine, basswood (*Tilia* spp.), quaking aspen, bigtooth aspen (*Populus grandidentata*), scarlet oak, chestnut oak (*Quercus prinus*), and northern red oak.

Pardo et al. (2011b) suggested that tree growth might be expected to decline at higher rates of N supply on nutrient-poor sites. Pardo et al. (2011b) reported ranges of CL of nutrient-N for northern hardwood and coniferous forests, including cover of herbaceous plants, lichens, and ectomycorrhizal fungi, in the Northern Forests ecoregion as follows:

Increased NO_3^- leaching in surface water	8 kg N/ha/yr	Aber et al. (2003)
Tree growth and mortality	10-26 kg N/ha/yr	McNulty et al. (2005)
Decreased growth and survivorship of northern forest tree species	3 kg N/ha/yr	Thomas et al. (2010)
Cover of herbaceous plant species	7-21 kg N/ha/yr	Hurd et al. (1998)
Community composition of lichens	4-6 kg N/ha/yr	Geiser et al. (2010)
Ectomycorrhizal fungal community structure	5-7 kg N/ha/yr	Lilleskov et al. (2008)
Arbuscular mycorrhizal fungal community composition and biomass	12 kg N/ha/yr	Van Diepen (2008)

Based on these data, the lowest CL reported by Pardo et al. (2011b) for vascular plants in the hardwood and coniferous forests of the Northern Forests ecoregion is 3 kg N/ha/yr for protecting against decreased tree growth and increased mortality of multiple tree species (Thomas et al. 2010). Pardo et al. (2011b) reported CL of 5-12 kg N/ha/yr for protecting mycorrhizal fungi (Lilleskov et al. 2008, Van Diepen 2008) in this ecoregion. Other studies suggested somewhat higher CL for protecting tree growth and mortality (10-26 kg N/ha/yr; McNulty et al. 2005), herbaceous plant cover (7-21 kg N/ha/yr; Hurd et al. 1998), and NO_3^- leaching from forested ecosystems (8 kg N/ha/yr; Aber et al. 2003).

Estimates reported by Pardo et al. (2011b) were considered reliable for protecting forests against nitrate (NO_3^-) leaching and fairly reliable for protecting tree growth and survivorship, loss of prominent herbaceous species, and change in ectomychorrizal fungal community structure. Results were less certain, and were based only on expert judgment, for predicting change in lichen community composition and biomass decline or for predicting community composition change of arbuscular mycorrhizal fungi (2011b).

Great Lakes Network
Dose Response and Critical Loads

Common vegetation types within Voyageurs that are considered potentially sensitive to nutrient-N enrichment effects attributed to atmospheric N deposition include forests, wetlands, and mycorrhizal fungi (Pardo et al. 2011b). Available information regarding CL estimates for each of these vegetation types is discussed in the sections that follow. Based on the analysis of Thomas et al. (2010), the CL for protecting forest ecosystems in Voyageurs were assumed to be as low as 3 kg N/ha/yr for protecting forest trees, 7 kg N/ha/yr for protecting cover of herbaceous plant species (Hurd et al. 1998), and 5 kg N/ha/yr for protecting ectomycorrhizal community structure (Lilleskov et al. 2008).

Forests

Map 34 shows the distribution of forest communities in Voyageurs NP that are expected to contain the N-sensitive tree species identified by Thomas et al. (2010). These forest types that are thought to be sensitive to nutrient-N enrichment in this network, based on the work of Thomas et al. (2010) occur at scattered locations throughout much of Voyageurs NP.

Wetlands

Wetlands are common throughout Voyageurs NP (Map 35). They contribute to high production of organic matter which is reflected in high dissolved organic carbon (DOC) concentrations in many lakes. Wetlands considered sensitive to N deposition typically contain plant species that have evolved under N-limited conditions. It is believed that the balance of competition among plant species in some sensitive wetland ecosystems can be altered by N addition, with resulting displacement of some species by others that can utilize the excess N more efficiently (U.S. EPA, 1993). This effect has been well documented in heathlands in The Netherlands, where common heather (Calluna vulgaris) has been replaced by grass species, presumably in response to atmospheric N deposition (Heil and Bruggink 1987, Tomassen et al. 2003). It is important to note, however, that N deposition in the Netherlands has been much higher (about 20 to 60 kg N/ha/yr) than levels commonly encountered in wetland areas in Voyageurs (less than about 6 kg N/ha/yr).

There is a great deal of diversity in terms of types of wetlands and how they respond to N addition. Bogs are often acidic and dominated by mosses. They are especially common in northern boreal forested regions such as are common in Voyageurs. They develop where precipitation is higher than evapotranspiration water loss in areas that exhibit an impediment to downward drainage in the soil (Mitsch and Gosselink 2000). Freshwater marshes develop where water inputs in groundwater plus surface water inflow approximate precipitation input (Koerselman 1989). Their vegetation is primarily tall graminoid plants. Freshwater swamps have hydrological conditions that are generally similar to marshes, but vegetation is forested (Greaver et al. 2011).
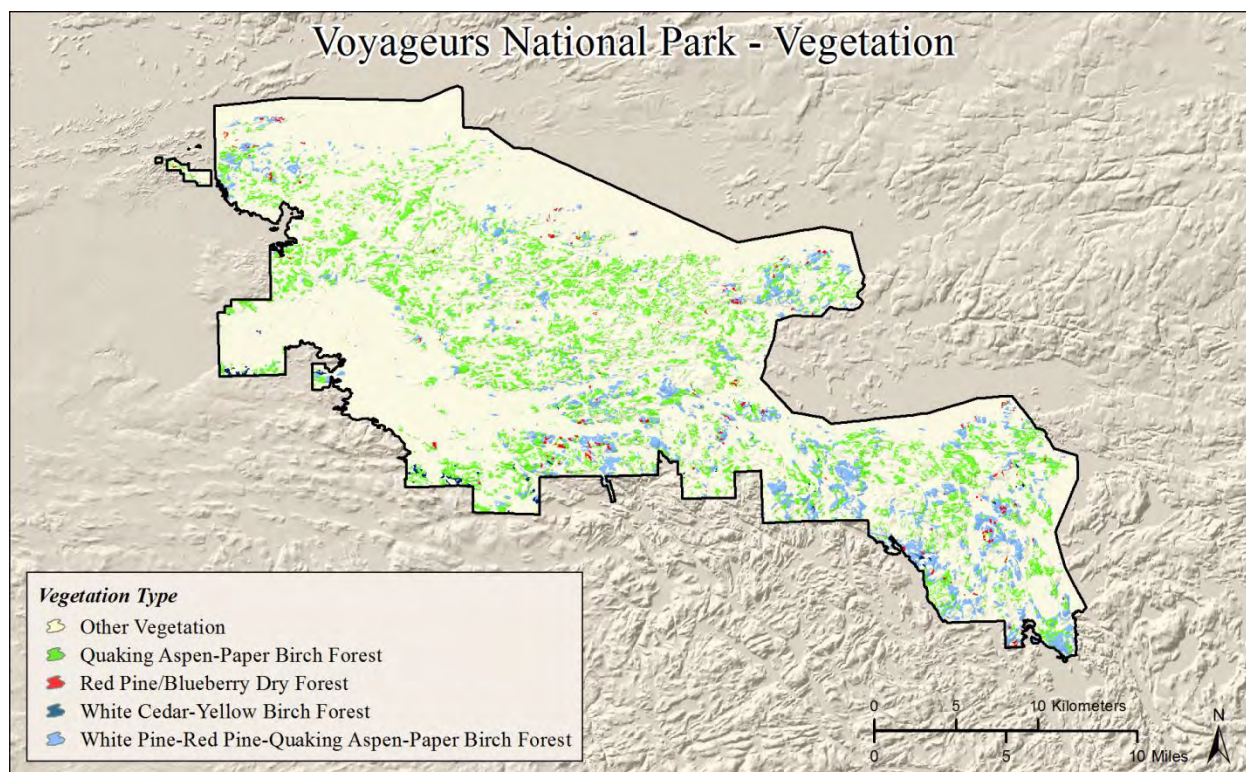
Map 34. Spatial distribution of N-sensitive forest communities in Voyageurs expected to contain red pine, red spruce, northern white cedar, yellow birch, eastern white pine, basswood, quaking aspen, bigtooth aspen, scarlet oak, chestnut oak, and/or northern red oak. These are the tree species in the Northern Forests ecoregion expected to be most sensitive to atmospheric N deposition (Thomas et al. 2010).
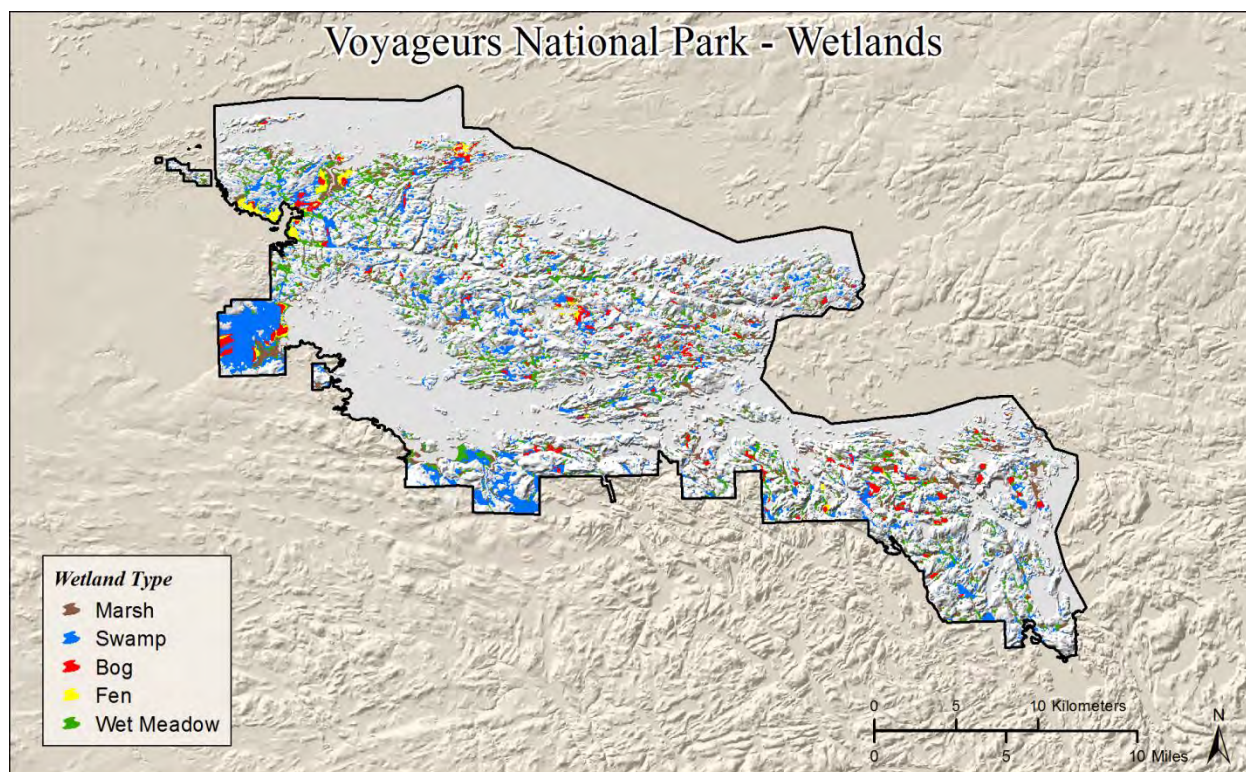
Map 35. Spatial distributions of wetland plant communities in Voyageurs National Park.

The response of freshwater wetlands to N addition varies with hydrological conditions. Bogs that receive much of their water input from precipitation are probably most sensitive to the effects of N input (Morris 1991). Most of their nutrient supply may be derived from precipitation and as a consequence plants are adapted to low inputs of N and other nutrients (Shaver and Melillo 1984, Bridgham et al. 1995). Bogs can host several federally listed rare and endangered plant species, including multiple species of quillworts (Isoetes spp.), sphagnum mosses (Sphagnum spp.), and the green pitcher plant (Sarracenia oreophila; Greaver et al. 2011).

Nutrient concentrations in wetland waters suggest that algal productivity in coastal Great Lakes wetlands may be N-limited. Historically, freshwater ecosystems, including the open-waters of the Great Lakes themselves, have been considered phosphorus (P)-limited (Hutchinson 1971, Rose and Axler 1998). However, Hill et al. (2006) found that more wetlands were N- than P-limited at each of the five Laurentian Great Lakes, and Morrice et al. (2004) measured a low ratio of N to P in Lake Superior coastal wetlands, also suggesting N-limitation. These results are consistent with the apparent general N-limitation of North American marsh lands (Bedford et al. 1999), and may result from differences in nutrient cycling in wetlands as compared to open-water ecosystems (Morrice et al. 2004). In wetlands, sediments tend to be oxygen-poor and organic C oxidation by primary producers in anoxic environments can be limited by the availability of NO_3- for use as an electron acceptor in the photosynthetic process (Sundareshwar et al. 2003).

Increased N availability in nutrient-poor wetland environments may lead to a decrease in species diversity and increased risk of extinction for some of the more sensitive and rare species (Moore et al. 1989). Peatlands and bogs are among the most vulnerable transitional ecosystems to adverse nutrient-enrichment effects of N deposition (Krupa 2003). The sensitivity of peatland Sphagnum species to elevated atmospheric N deposition is well documented in Europe (Berendse et al. 2001, Tomassen et al. 2004). Sphagnum squarrosum and S. fallax have been observed to be negatively affected by experimentally elevated atmospheric N and sulfur (S) inputs in Europe (Kooijman and Bakker 1994). Roundleaf sundew (Drosera rotundifolia) is also susceptible to elevated atmospheric N deposition (Redbo-Torstensson 1994).

In other studies, wetland species such as Calluna vulgaris can successfully compete with grasses even at relatively high rates of N deposition, as long as the vegetative canopies are closed (Aerts et al. 1990). However, N deposition causes nutrient imbalances, including increase in the shoot-to-root ratio, and therefore increases in the sensitivity of shrubs to drought stress, frost stress, and attack by insect pests (Heil and Diemont 1983). These can result in gaps in the canopy of the shrub layer, which can then be readily invaded by grasses that are more efficient in using the additional N and therefore gain a competitive advantage (Krupa 2003).

Data are not available with which to evaluate the extent to which wetlands in Voyageurs have been affected by nutrient enrichment from N deposition. Wetlands that are widely distributed, receive moderate levels of N deposition. Wetlands that receive moderate levels of N deposition are widely distributed within the Great Lakes Network, including within Voyageurs NP. The levels of N deposition commonly found in portions of Voyageurs NP that contain appreciable wetland cover may or may not be sufficiently high so as to cause species shifts in wetland plants. If such effects do occur, they are most likely in wetlands such as bogs and poor fens that normally receive most of their nutrients from atmospheric inputs. These wetlands have been shown in Europe to experience changes in plant species composition in response to high levels of atmospheric N deposition. It is not clear to what extent such effects occur under ambient N deposition levels in Voyageurs NP. The risk of species composition change is important, in part because wetland ecosystems often contain relatively large numbers of rare plant species.

Greaver et al. (2011) estimated that the CL of atmospheric N deposition to protect peatlands from increased productivity was in the range of 2.7 to 13 kg N/ha/yr. This estimate was based on consideration of the results of the studies of Aldous et al. (2002), Moore et al. (2004), Rochefort et al. (1990), and Vitt et al. (2003). Regional estimates of atmospheric N deposition suggest that wetlands in Voyageurs NP likely receive more than the lower CL level (2.7 kg N/ha/yr; Map 36) estimated by Greaver et al. (2011).
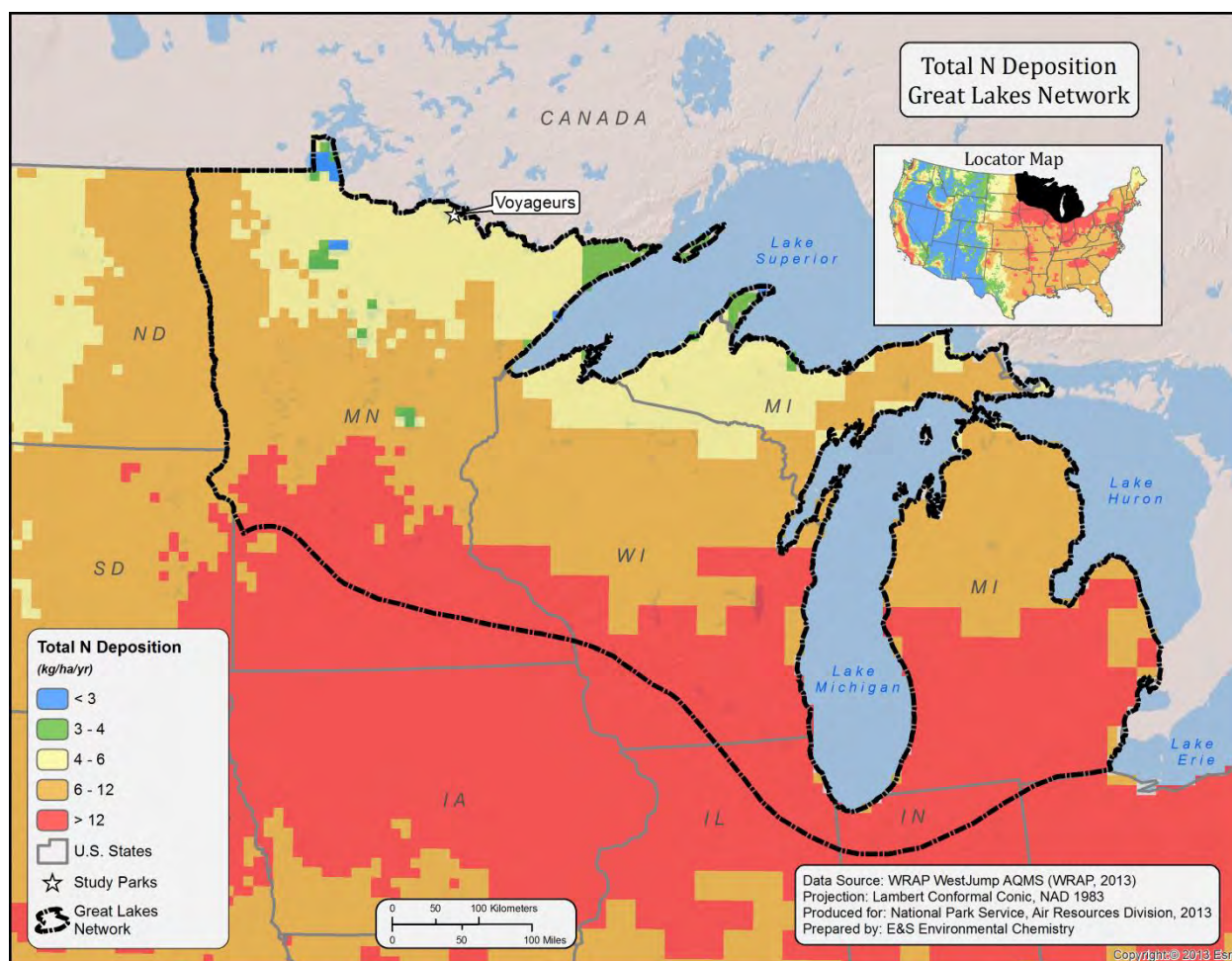
Map 36. CAMx model estimates of total wet plus dry N deposition (kg/ha/yr) for the year 2008 throughout the Great Lakes Network.

Grasslands

Atmospheric deposition of N to grassland communities can cause changes in plant species distribution and abundance, including reduced species richness and biodiversity. However, potentially N-sensitive grassland vegetation only covers 2.3% of Voyageurs NP. There are not data available for specification of an appropriate CL for protecting grassland communities in the Northern Forests ecoregion. Experimental work to date has been based on N fertilization at levels much higher than ambient N deposition.

Mycorrhizal Fungi

Lilleskov et al. (2008) found that community structure of ectomycorrhizal fungi in northern forests changed over a regional N deposition gradient from northeastern New York to Maine. Frequency of the different morphotypes varied continuously with increasing root N. Root N was positively related to N deposition. Wet deposition at the low N end of this gradient was estimated at approximately 3 kg N/ha/yr; at this level of wet deposition, total N deposition would likely be approximately 4 kg

51

N/ha/yr (Ollinger et al. 1993). The estimated level of N deposition resulting in mycorrhizal fungi community change is 5 to 7 kg N/ha/yr. This is quite similar to levels estimated for mycorrhizae change in white spruce (*Picea glauca*) forests along a depositional gradient in Alaska (Lilleskov et al. 2001, Lilleskov et al. 2002) and pitch pine (*Pinus rigida*) forests in the Pine Barrens of New Jersey (Dighton et al. 2004). If elevated soil NO_3^- availability is a good indicator of ectomycorrhizal fungal community change in oligotrophic N-limited conifer forests (Lilleskov et al. 2002, Lilleskov 2005), then N deposition thresholds that minimize excess NO_3^- availability should be sufficient to protect ectomycorrhizal fungal diversity in these forest types.

Less information is available on arbuscular mycorrhizal community response to N in northern forests. One study in sugar maple-dominated forests in Michigan found declines of arbuscular mycorrhizal fungal biomass (van Diepen et al. 2007, 2010) and changes in community structure (Van Diepen 2008) in response to N addition of 30 kg/ha/yr above background deposition of approximately 5 to 12 kg N/ha/yr wet deposition, but the lower threshold of response to N is not known (Clark 2011). There is some indication that arbuscular mycorrhizal biomass has declined and community composition has shifted over the N deposition gradient (Van Diepen 2008), suggesting a threshold of <12 kg N/ha/yr of wet deposition, but it is likely that there are interactions with site factors, and definitive determination of a response threshold has not been made (Clark 2011).

Emissions and Deposition
County-level NO_x emissions data are available for GLKN from the NEI, but no additional work has been done by WestJump to include more of the small N emissions sources in this network. Therefore, maps of oil and gas emissions are not shown for this network. Total NO_x emissions from all source types are relatively high (> 25,000 tpy) at several locations in this network, in particular around Chicago and Detroit (Map 37).

Map 36 shows CAMx model estimates of total wet plus dry N deposition at 36 km resolution across the Great Lakes Network in the year 2008. Ambient deposition below 3 kg N/ha/yr is shown in blue. All colors other than blue reflect portions of the network for which ambient N deposition exceeds this lowest CL of 3 kg N/ha/yr determined by Thomas et al. (2010) to protect against decreased growth and survivorship of several tree species in this region. Atmospheric deposition of N in this region is due largely to agricultural sources of reduced N and urban sources of oxidized N. The region in and around Voyageurs NP receives an estimated total N deposition in the range of 4-6 kg N/ha/yr.
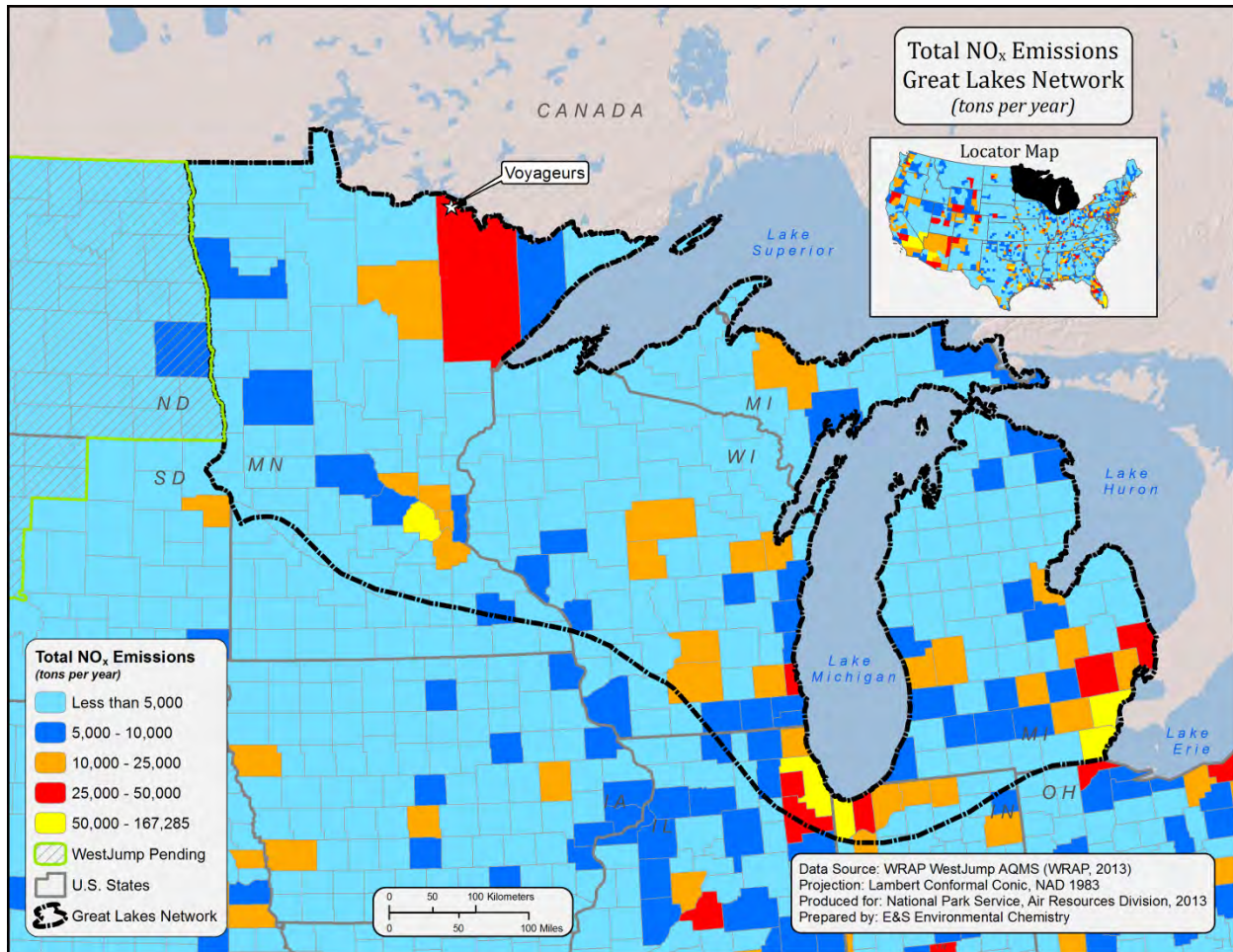
Map 37. Estimated NO_x emissions from all sources (tpy) in the Great Lakes Network for the year 2008.

Exceedance

A CL exceedance map for Voyageurs NP is shown in Map 38. Much of the park is covered by vegetation thought to be sensitive to nutrient-N enrichment. Essentially all of the identified N-sensitive plant community types in this park are in exceedance, based on Pardo et al.'s (2011b) lower limit of the empirical CL of 3 kg N/ha/yr and CAMx model projections of 4-6 kg N/ha/yr of ambient deposition. An analysis of exceedance based on Greaver et al.'s (2011) estimate of peatland CL of 2.7 kg N/ha/yr would yield similar results.
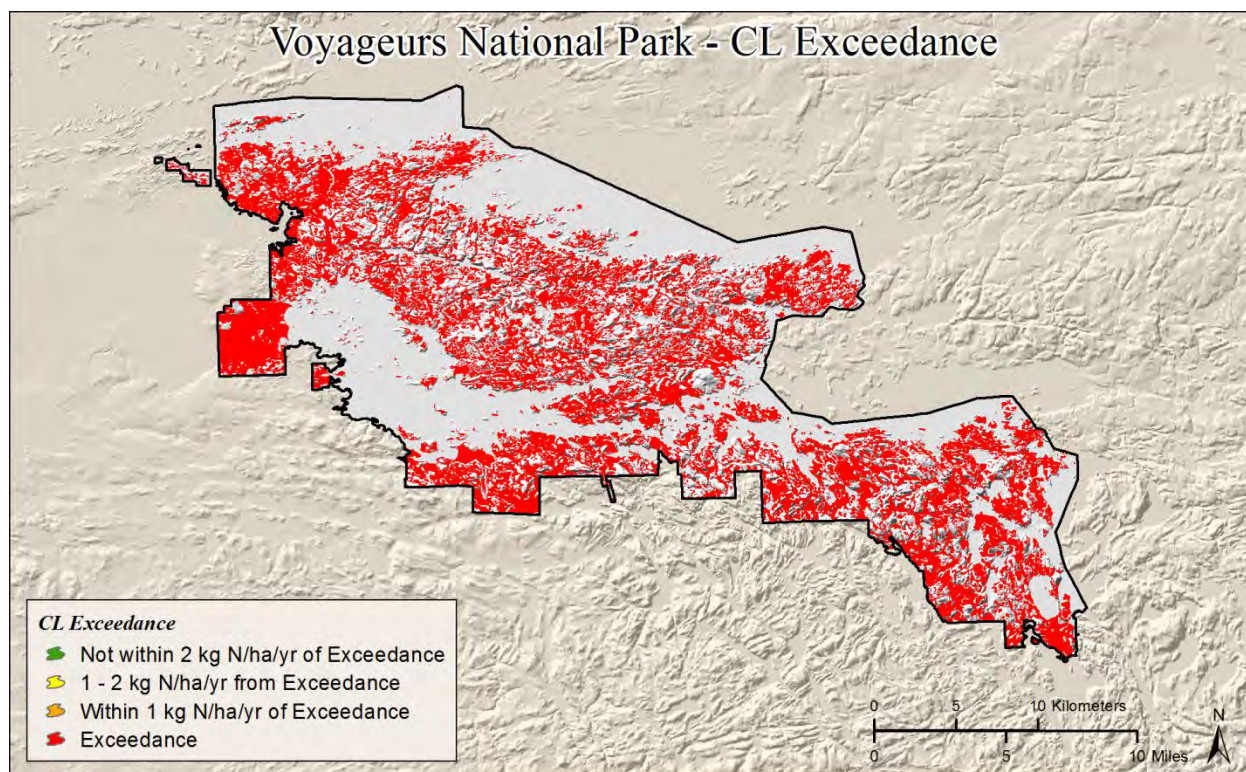
Map 38. Locations of vegetation types in Voyageurs National Park that exhibit CL exceedance due to nutrient-N enrichment. The area of sensitive vegetation is labeled red if total N deposition in year 2008 is in exceedance of the lower limit of the critical load (CL; 3 kg N/ha/yr) estimated by Pardo et al. (2011b) for the sensitive vegetation types found in the park. The remaining areas of sensitive vegetation are classified according to the extent to which total N deposition is below the CL. Areas showing the background color do not contain vegetation types identified by Pardo et al. (2011b) as being highly sensitive to nutrient-N enrichment.

Great Plains Ecoregion
Northern Great Plains Network
Dose Response and Critical Loads

Research studies conducted on the Great Plains (and supported by studies in Europe) have shown that excess N can alter grassland biodiversity, promote growth of non-native grasses, enhance insect infestation, and promote loss of grassland ecosystems at landscape scales by favoring encroachment of trees and shrubs. Many of these studies were considered in the review of Pardo et al. (2011b). Much of the work on nutrient enrichment effects of N deposition on grasslands has been conducted in Europe. Nitrophilous plant species have been shown to have increased in abundance, and N-sensitive species to have declined in European grasslands over approximately the last half of the 20th century (Bobbink et al. 1998). It is believed that these changes were caused, at least in part, by atmospheric N deposition. Recent data were compiled for the review of Pardo et al. (2011a) and are summarized below. The clearest evidence that atmospheric N deposition over a large region has actually impacted terrestrial biodiversity was provided by Stevens et al. (2004), who reported results of 68 plot (4 m^2) surveys across a range of atmospheric N deposition from 5 to 35 kg N/ha/yr.

Species richness in grasslands was found to decline as a linear function of N deposition, with a reduction on average of one plant species for every 2.5 kg N/ha/yr of N input.

Nutrient-N enrichment can alter the species composition and biodiversity in grassland communities in the Great Plains. Grasslands in the northern Great Plains are mainly N-limited or co-limited by N, P and/or water. Many grassland species respond rapidly to changing environmental conditions (Vitousek and Howarth 1991, Hooper and Johnson 1999, Knapp and Smith 2001, Clark 2011). It is difficult to ascertain the likely effects of N addition to these ecosystems without also considering the effects of periodic disturbance (especially fire, drought, and herbivory; Seastedt and Knapp 1993, Knapp et al. 1998).

Many studies of the effects of N enrichment on grasslands of the Great Plains have involved experimental addition of very high amounts of N (> 30 kg N/ha/yr; Clark 2011). Some experiments included lower levels of N addition to grassland ecosystems, mainly in Minnesota. For example, Tilman (1987) added N to four sites in Minnesota at levels ranging from 10 to 270 kg N/ha/yr. At the lower rates of experimental N addition (10 to 20 kg N/ha/yr) to two old fields, much of the added N was retained in the soil after 22 years of treatment (Wedin and Tilman 1996). Nevertheless, the N retention efficiency decreased, soil extractable NO_3^- increased, and plant species composition shifted over time in response to N addition (Clark and Tilman 2008, Clark et al. 2009).

Reduced species richness and biodiversity in response to N addition are of concern for grassland communities, in large part because of the potential interactions among diversity and ecosystem processes and functioning. It is more likely that a species-poor plant community may exhibit compromised ecosystem functioning as compared with a species-rich community. There are also interactions with the availability of other resources. For example, shortgrass prairie is considered to be less sensitive to N addition than tallgrass prairie because the former is more likely to experience water limitation (Lauenroth et al. 1978, Hooper and Johnson 1999, Clark 2011).

In tallgrass prairie, C_3 grasses such as *Elymus virginicus* and *E. canadensis* showed a greater positive growth response to N additions than C_4 grasses (*Andropogon geradii*, *Schizachyrium scoparium*) and forbs (*Solidago nemoralis*, *S. rigida*; Lane and BassiriRad 2002). Species with smaller initial biomass exhibited the greatest increase in biomass, with a sevenfold to eightfold increase in *S. nemoralis* and *E. canadensis* and only a threefold increase in *S. rigida* (Lane and BassiriRad 2002). In experiments where N fertilization to simulate increased N deposition was applied to common ragweed (*Ambrosia artemisifolia*), simulated N deposition led to increased vegetative and seed biomass and decreased root: shoot ratios (Throop 2005).

In another study, experimental N addition for 12 years to two Minnesota old fields at levels of 10 and 20 kg N/ha/yr decreased the abundance of four grasses by almost 20%, and increased the abundance of invasive Kentucky bluegrass (*Poa pratensis*) and quackgrass (*Agropyron repens*; Knops and Reinhart 2000). Losses of grass species and changes in species composition generally increased over time with experimental N addition, especially at lower addition levels (Clark 2011). After 22 years of N addition at a level of 10 kg N/ha/yr, the number of species declined by 17% (Clark and Tilman

2008). Tilman (1993) hypothesized that species loss was caused mainly by increased litter mass, although light limitation from competing plants was also likely important (Hautier et al. 2009).

Experimental N addition at > 50 kg N/ha/yr to tallgrass prairie increased productivity and decreased species richness, probably as a consequence of reduced light penetration and increased litter biomass (Baer et al. 2003). Wedin and Tilman (1996) presented results of 12 years of experimental N addition to 162 grassland plots in Minnesota. N loading dramatically changed plant species composition, decreased species diversity, and increased aboveground productivity in their treatment plots. Species richness declined by more than 50% across the N-deposition gradient, with the greatest losses at levels of N input of 10 to 50 kg N/ha/yr. This loss of species diversity was accompanied by large changes in plant species composition, with C_4 grasses declining and the weedy Eurasian C_3 grass quackgrass becoming dominant at high N addition rates (Wedin and Tilman 1996). The authors concluded that N loading is a major threat to grassland ecosystems and causes loss of diversity, increased abundance of non-native species, and the disruption of ecosystem functioning. A major uncertainty, however, is the rate of N loading at which such changes may be manifested. There is relatively little information on the effects of low levels of N addition from the atmosphere to prairie ecosystem in NGPN.

Effects of N fertilization, in turn, can influence faunal populations. For example, fertilization in a Minnesota old-field at moderate rates (≤ 20 kg N/ha/yr) for 14 years resulted in decreased species diversity of insects in response to decreased diversity of plant food sources. Plant species richness decreased, and quackgrass and Kentucky bluegrass became dominant in response to the N addition (Haddad et al. 2000). Changes in the abundance of insect functional groups were also observed. Herbivores (especially the dominant species) increased in numbers; parasitoid insect species decreased. Over the long term, changes in plant species composition would be expected to either increase or decrease insect herbivore activity, depending on whether there is a shift toward or away from herbivore-preferred plant species (Throop et al. 2004, Clark 2011).

Increased availability of N to grasses can affect herbivores that feed on grasses by altering food quality, quantity, and phenology, and also perhaps by changing the relationships between herbivores and their predators (Throop and Lerdau 2004). Nitrogen fertilization at very high rates of 54 and 170 kg N/ha/yr (as NH_4NO_3) led to a decline in plant species composition in an oak savanna site (Avis et al. 2003). In the control plots, five species collectively accounted for more than 40% of the plant cover versus four plant species in the lower N addition plots. In the higher N addition plots, a single plant species accounted for more than 40% of the plant cover.

Experimental N addition for five years at a rate of 16.3 kg N/ha/yr to a mixed-grass prairie in Oklahoma caused an increase in soil NO_3^-; leaching of N increased about 12-fold (Jorgensen et al. 2005, Clark 2011). Total plant biomass increased, and the cover of tall fescue (*Schedonorus phoenix*) increased twofold (Clark et al. 2003, Jorgensen et al. 2005). Shortgrass prairie is considered to be less sensitive to N addition than tallgrass prairie because the former is more likely to experience water limitation (Lauenroth et al. 1978, Hooper and Johnson 1999, Clark 2011).

Grasslands of the Great Plains have also experienced invasions of woody plant species (Kleb and Wilson 1997) that appear to be driven, at least in part, by N enrichment of soil (Tilman 1987). Fertilization may increase the water use efficiency of woody species (Bert et al. 1997), and enable them to colonize temperate grassland sites (Köchy and Wilson 2001). Accelerated rates of N cycling (Carreiro et al. 2000) may increase competition for light (Wilson and Tillman 1991) and give advantage to woody plants (Aerts et al. 1999, Köchy and Wilson 2001). High N deposition has been shown to be correlated with an increase of tall plant species in nutrient-poor European grasslands (Bobbink et al. 1998). Köchy and Wilson (2001) quantified forest invasion of aspen trees into grasslands in six national parks in western Canada that received a range of levels of atmospheric N deposition. The rate of aspen expansion was measured from aerial photographs taken over a period of six decades. Forests in high-deposition parks expanded 10 times faster than forests in low-deposition parks. The average forest expansion rate was 1% per year in high-deposition parks (Köchy and Wilson 2001).

Three parks within the Northern Great Plains Network were included in the I&M vegetation mapping effort: Theodore Roosevelt NP, Badlands NP, and Wind Cave NP (Cogan et al. 1999, Von Loh et al. 1999, Von Loh et al. 2000). These are the parks assessed here. Grassland communities in these parks labeled by USGS and NPS as medium-tall were mapped for this study as having CL appropriate to tallgrass prairie as estimated by Clark (2011). This was done in order to err on the side of resource protection by assigning the lower CL range of tallgrass prairie (5-15 kg N/ha/yr) to these communities rather than the CL range of the less sensitive shortgrass or mixed-grass prairie (10-25 kg N/ha/yr).

Badlands National Park

The dominant vegetation of Badlands NP is a moderately dense short to medium tallgrass prairie, although this designation encompasses a variety of grassland communities (Cushman and Jones 1988, Froiland and Weedon 1990). These in-park prairies constitute important AQRVs that are sensitive to nutrient enrichment impacts from atmospheric N deposition. They are mainly dominated by western wheatgrass (*Agropyron smithii*), green needlegrass (*Stipa viridula*), blue grama (*Bouteloua gracilis*), and needle-and-thread (*Stipa comata*). Associated species include fringed sage (*Artemisia frigida*), prairie junegrass (*Koeleria pyramidata*), little bluestem (*Andropogon scoparius*), silky wormwood (*Artemisia dracunculus*), purple coneflower (*Echinacea angustifolia*), and various other forbs. Blue grama and buffalo grass (*Bouteloua dactyloides*) form a mosaic of patches in combination with western wheatgrass and green needlegrass. The vegetation of Badlands NP is an example of these highly valued mixed and tallgrass prairie ecosystems, which have largely been extirpated from their former ranges.

Von Loh et al. (1999) identified vegetation map units in Badlands NP, based on aerial photographs. Data from 137 observation points, selected to sample the range of habitat and vegetation variability, were collected during field surveys to verify and refine the aerial interpretation. Of the 33 vegetation map units developed in Badlands NP, we identified for the purposes of this report the following plant community types as belonging to prairie categories:

- shortgrass prairie
- blue grama grassland
- tallgrass prairie
- introduced grassland
- little bluestem-grama grasses – threadleaf sedge (*Carex filigolia*) grassland
- seeded mixed-grass prairie
- switchgrass (*Panicum virgatum*) grassland
- western wheatgrass grassland alliance
- western wheatgrass – green needlegrass grassland

Portions of Badlands NP covered by the tallgrass and shortgrass prairies listed above are shown in Map 39. These are the plant community types present in Badlands NP that are expected to be most sensitive to nutrient-N enrichment.
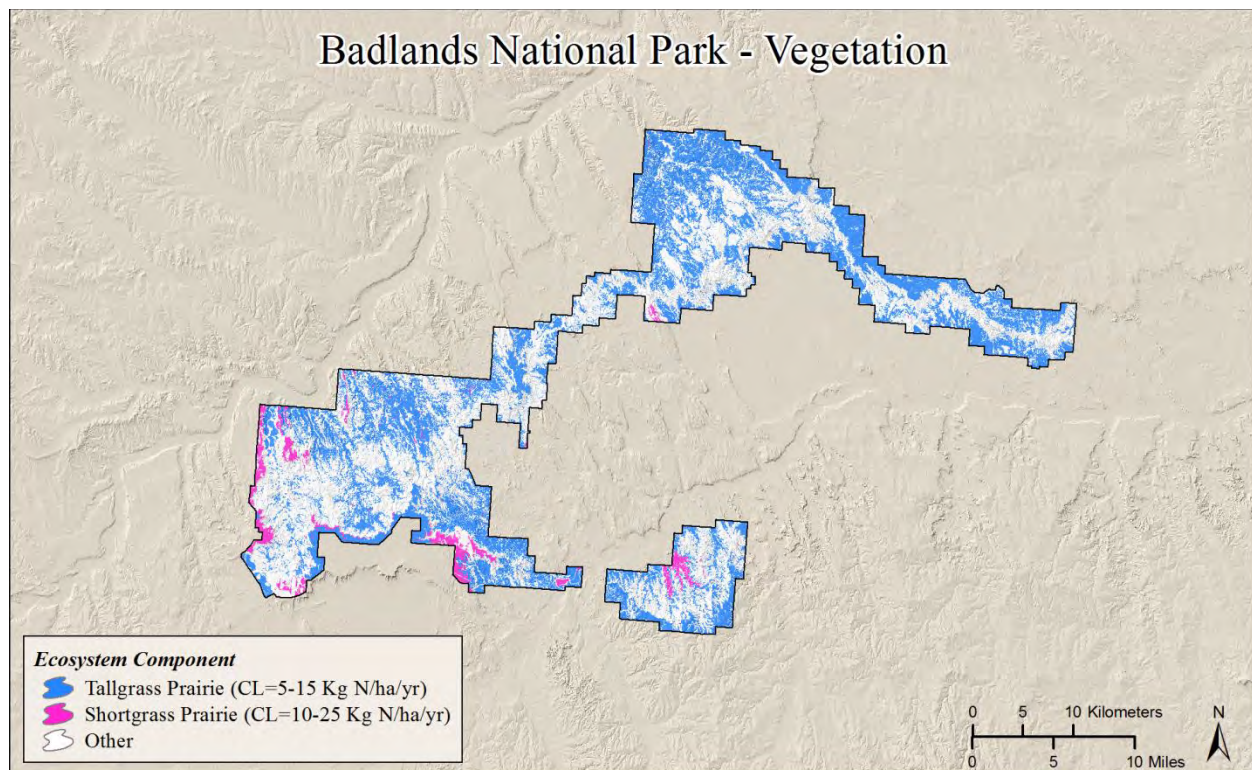
Map 39. Locations in Badlands National Park having vegetation types characterized by the presence of plant communities expected to be most sensitive to nutrient-N deposition, based on nutrient critical load values compiled by Clark (2011).

Several of the map units classified here as tallgrass prairie likely contain a mix of short and tallgrass species (i.e., the two western wheatgrass units and the seeded mixed-grass prairie unit). These map units are classified as the more sensitive tallgrass prairie type. Thus, they are considered to have CL in the low range (5-15 kg N/ha/yr) (Clark 2011) in order to err on the side of resource protection.

Theodore Roosevelt National Park

Von Loh et al. (2000) identified seven vegetation map units in Theodore Roosevelt NP that represent grassland types. Of those, the following can be classified as tallgrass prairies, or containing some elements of tallgrass prairie:

- introduced grassland
- little bluestem – sideoats grama (Bouteloua curtipendula) herbaceous alliance
- needle and thread herbaceous alliance
- prairie sandreed (Calamovilfa longifolia) herbaceous alliance
- seeded mixed-grass prairie
- western wheatgrass herbaceous alliance

Additionally, the following was classified as representing mixed-grass prairie:

- Prairie cordgrass (*Spartina pectinata*) temporarily flooded

Locations in Theodore Roosevelt NP having plant community types expected to be most sensitive to nutrient-N deposition are shown in Map 40. Tallgrass and shortgrass prairies have estimated nutrient-N critical loads of 5-15 and 10-25 kg N/ha/yr, respectively (Clark 2011).

Wind Cave National Park

Cogan et al. (1999) identified vegetation map units in Wind Cave NP, based on aerial photography. Field surveys were conducted at 134 observation points, selected to sample the range of habitat and vegetation variability in the park. The following vegetation map units in Wind Cave NP were identified as tallgrass prairie or as containing elements of tallgrass prairie:

- little bluestem – grama grass herbaceous vegetation
- little bluestem sideoats grama herbaceous alliance (with burned ponderosa pine)
- needle and thread – blue grama-threadleaf sedge herbaceous vegetation
- western wheatgrass – Kentucky bluegrass complex
- western wheatgrass – Kentucky bluegrass complex (with burned ponderosa pine)

Additionally, the following were classified as representing short or mixed-grass prairie:

- introduced weedy graminoid herbaceous vegetation
- purple three-awn fetid marigold herbaceous vegetation

Locations in Wind Cave NP having these plant community types expected to be most sensitive to nutrient-N deposition are shown in Map 41. Tallgrass and shortgrass prairies have estimated nutrient critical N loads of 5-15 and 10-25 kg N/ha/yr, respectively (Clark 2011).

Map 40. Locations in Theodore Roosevelt National Park having vegetation types characterized by the presence of plant communities expected to be most sensitive to nutrient-N deposition, based on nutrient critical load values compiled by Clark (2011).
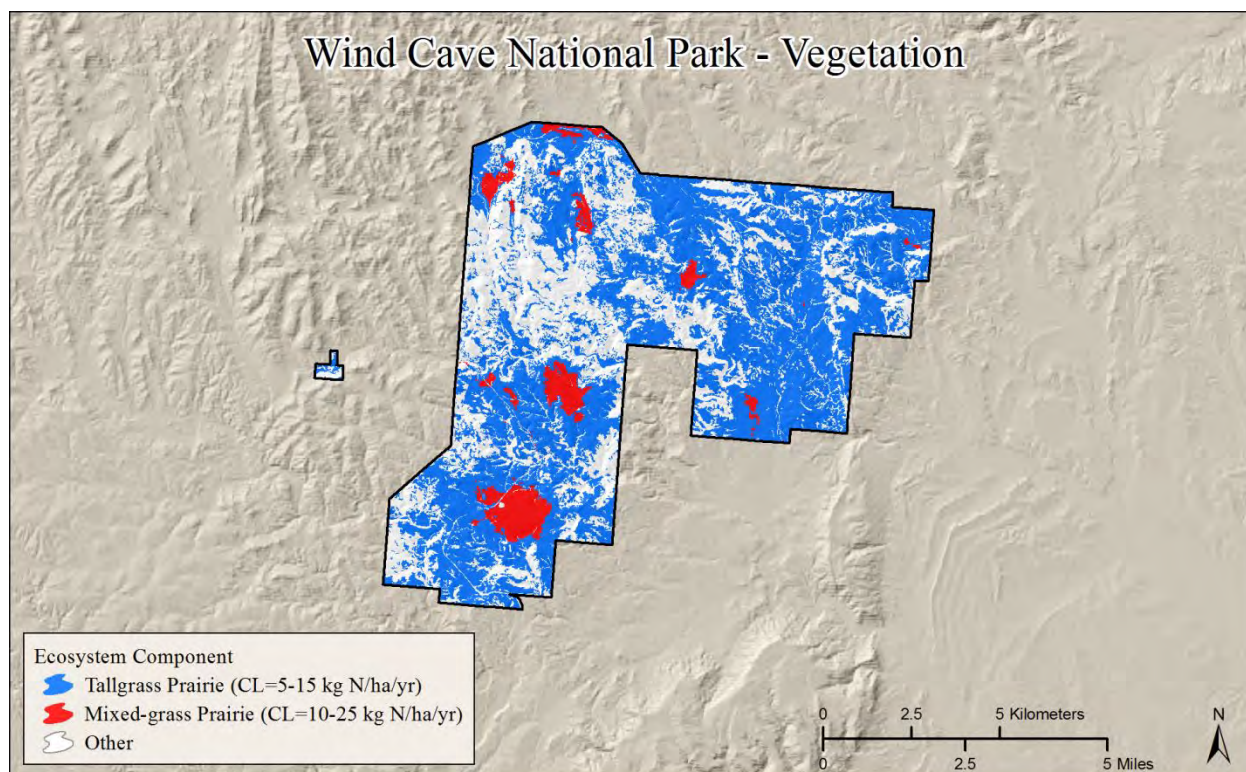
Map 41. Locations in Wind Cave National Park having vegetation types characterized by the presence of plant communities expected to be most sensitive to nutrient-N deposition, based on nutrient critical load values compiled by Clark (2011).

Emissions and Deposition

For much of the NGPN, WestJump emissions estimates are not yet available. The current rates of oil and gas related NO_x emissions data may be higher than what are shown here because this region has seen an increase in the amount of oil and gas exploration and production in recent years. WestJump data that are available cover the western network boundary and areas of Wyoming and Colorado to the west and southwest. Based on available data, some counties in proximity to Badlands NP and Wind Cave NP had 2008 oil and gas NO_x emissions higher than 1,000 tpy, and one county had more than 10 times that level (Map 42). Several of these counties emitted more than 1 ton/mi2/yr from oil and gas exploration and production (Map 43). Total NO_x emissions were elevated mainly to the west of Badlands NP and Wind Cave NP, in the range of 10,000 to 50,000 tpy from several counties located along the network boundary (Map 44). In these areas, oil and gas exploration and production constituted an appreciable percentage (in some cases > 60%) of the total NO_x emissions from all sources (Map 45).

Total N deposition in the NGPN region varied from west to east, with levels generally below 4 kg N/ha/yr in the west to greater than 6 kg N/ha/yr throughout the eastern and southern portions of the network region. Deposition in the vicinity of the NGPN parks has ranged between about 3 and 6 kg N/ha/yr based on CAMx estimates shown on Map 46.

61
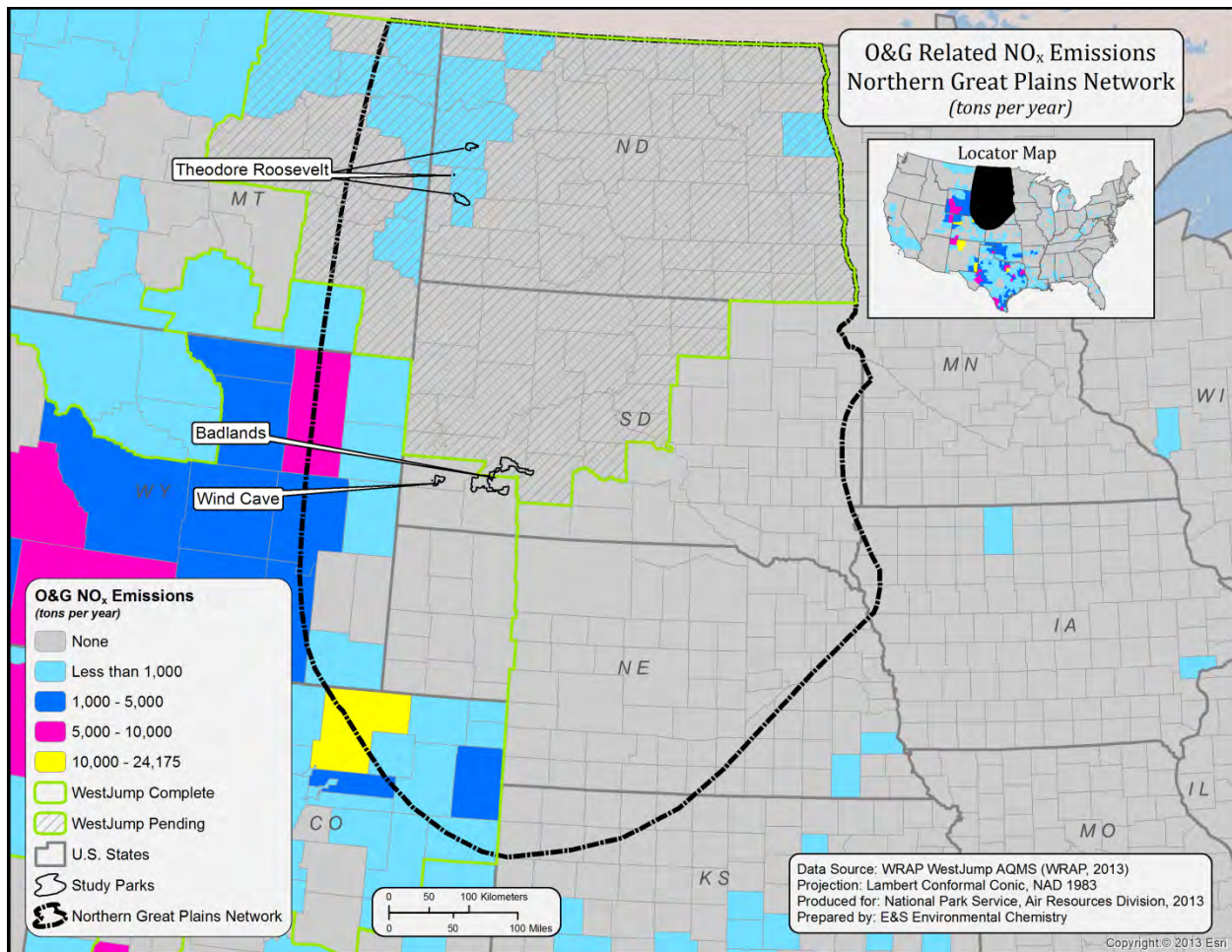
Map 42. Estimated oil and gas NO$_x$ emissions (tpy), by county, in the Northern Great Plains Network for the year 2008, which is the most recent year for which data are available at the time of this writing. Note that oil and gas emissions levels in some areas have likely increased in more recent years.
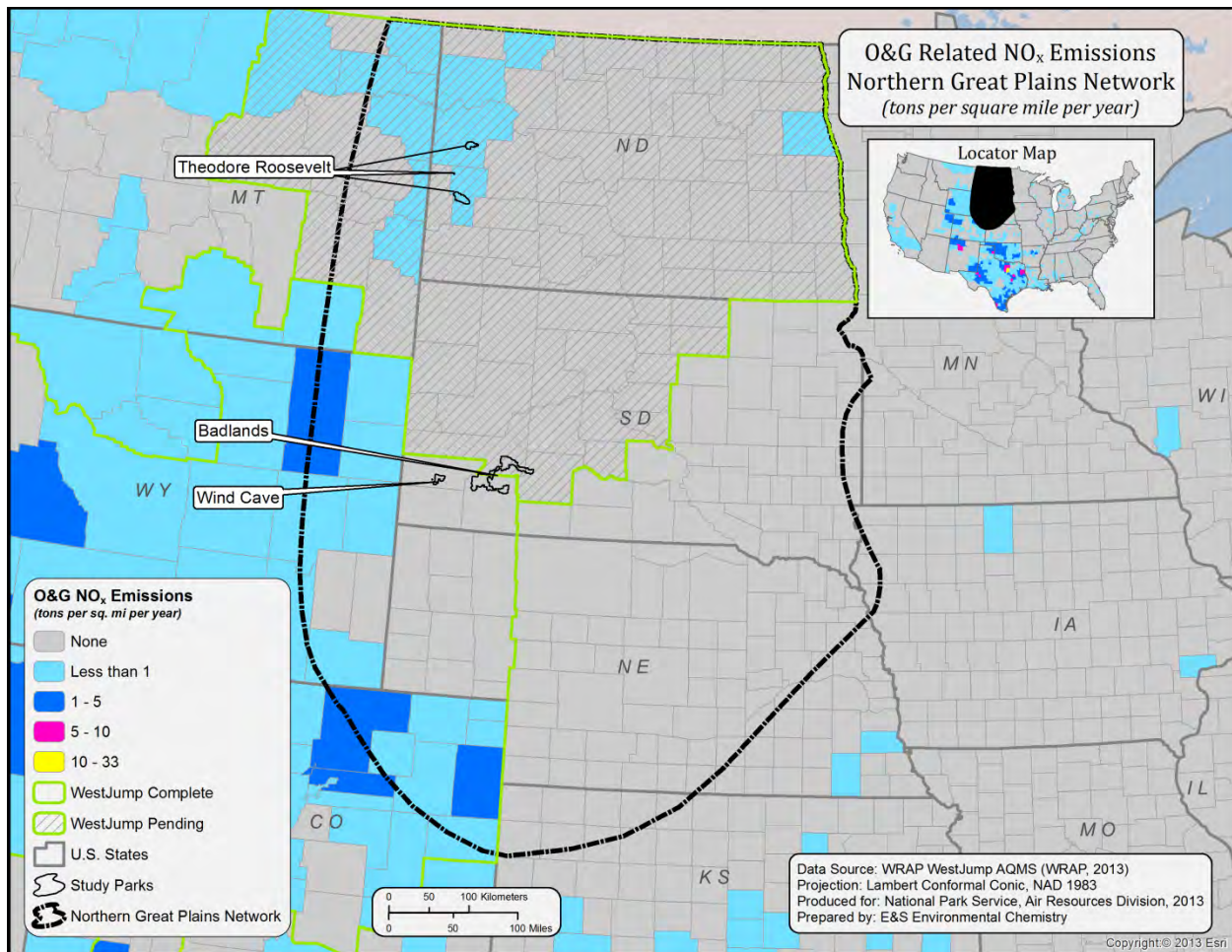
Map 43. Estimated oil and gas NO$_x$ emissions per unit land area (tons/mi^2/yr) in the Northern Great Plains Network for the year 2008.
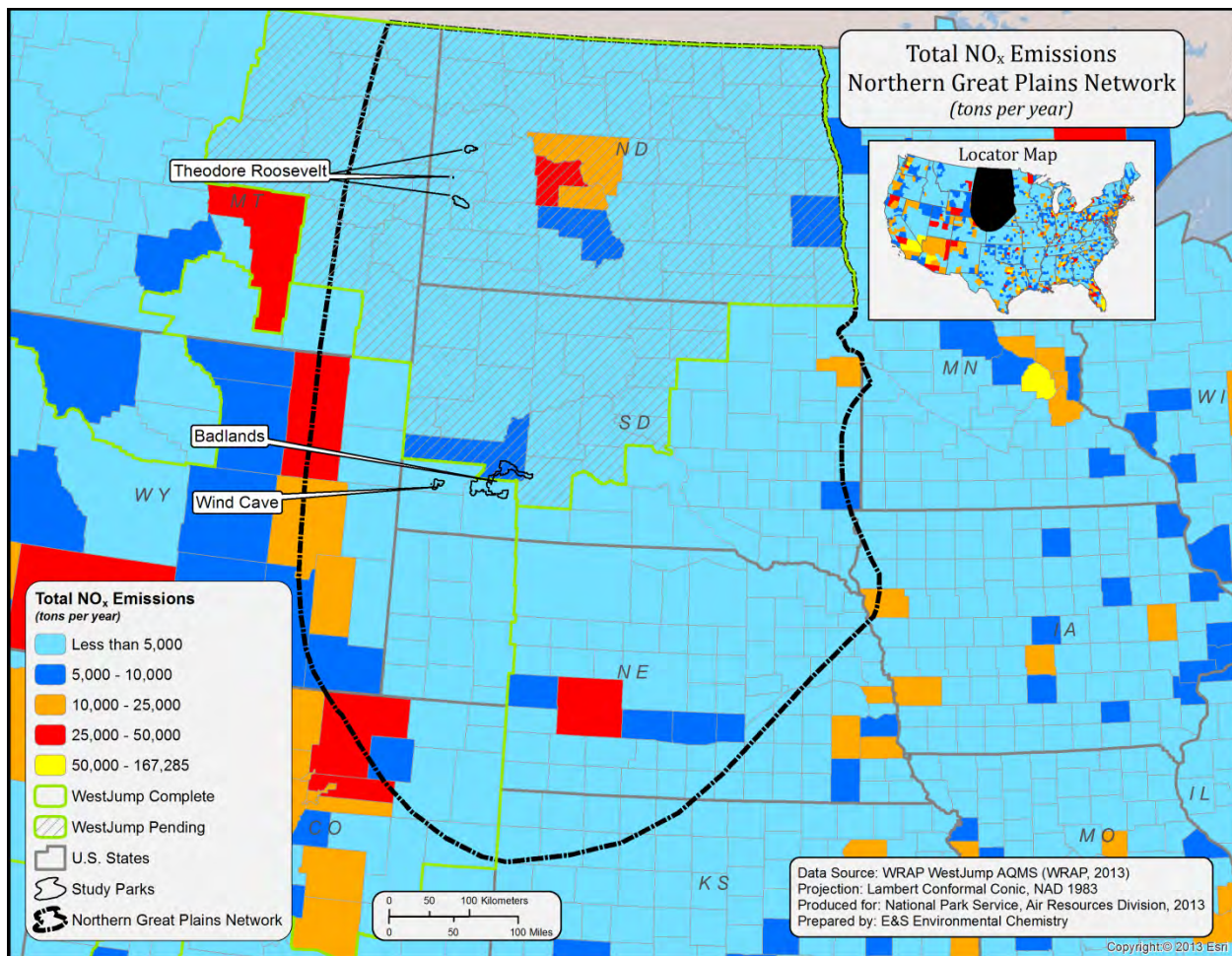
Map 44. Estimated NO_x emissions from all sources (tpy), including oil and gas, in the Northern Great Plains Network for the year 2008.
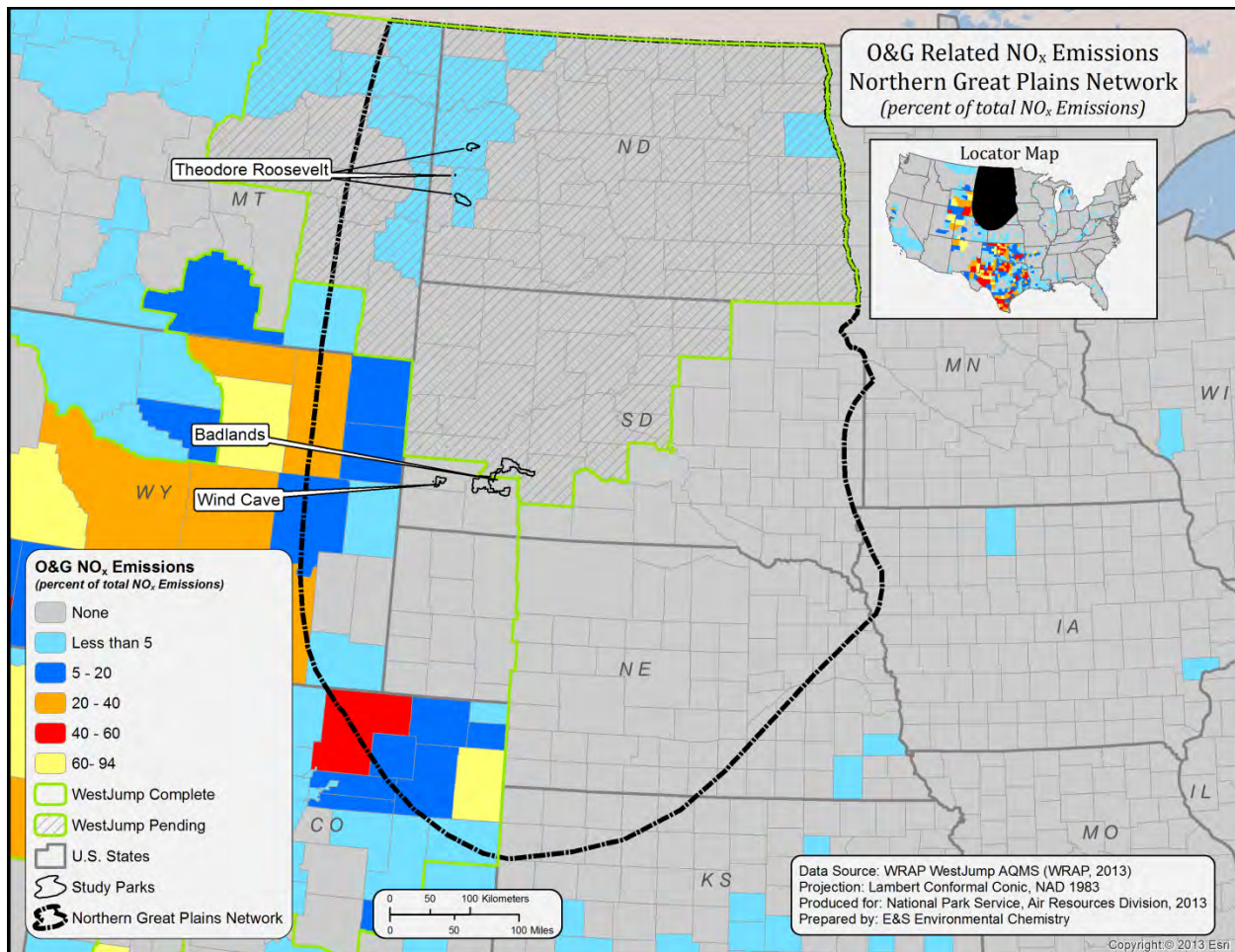
Map 45. Estimated oil and gas NO_x emissions in proximity to study parks in the Northern Great Plains Network as a percentage of total NO_x emissions for the year 2008.
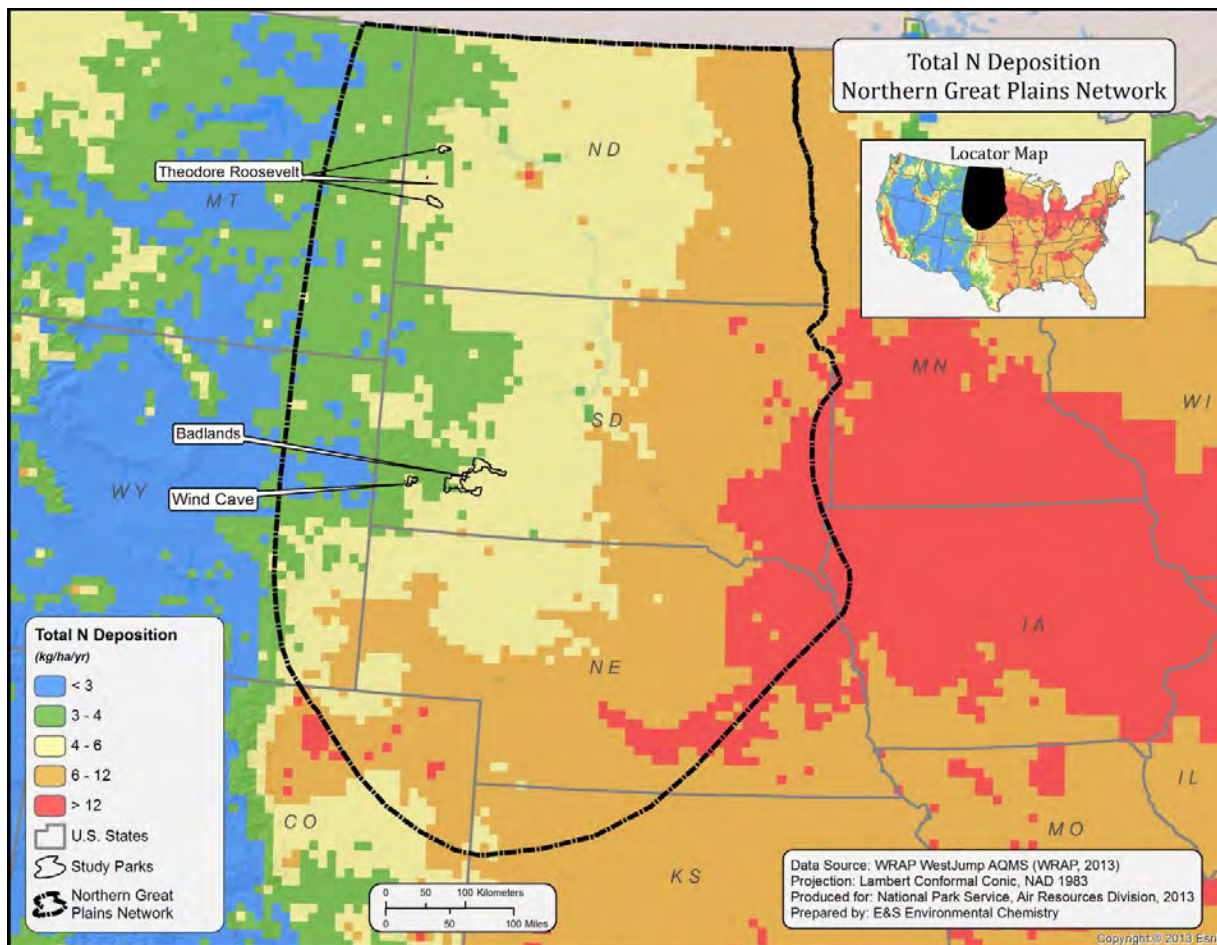
Map 46. CAMx model estimates of total wet plus dry N deposition (kg/ha/yr) for the year 2008 throughout the Northern Great Plains Network.

Exceedance

Based on CAMx modeling for 2008, only a small portion of easternmost Badlands NP receives ambient N deposition in exceedance of the nutrient-N CL for tall grass prairies (5-10 kg N/ha/yr) (Map 47). None of the park lands in Theodore Roosevelt NP or Wind Cave NP are in exceedance (Maps 48 and 49).

Nevertheless, more than half of Badlands NP, Wind Cave NP, and much of the southern unit of Theodore Roosevelt NP received ambient N deposition (in 2008) below, but within 1 kg N/ha/yr of, the lower CL estimate (5 kg N/ha/yr) compiled by Clark (2011). Thus, any appreciable increase in NO_x emissions from oil and gas exploration and production, or other sources, will increase the likelihood of exceedance and consequent ecological changes associated with excess nutrient-N supply. Given the incomplete nature of the oil and gas and agricultural emissions inventory for the Northern Great Plains, CAMx may be underestimating or overestimating current N deposition.
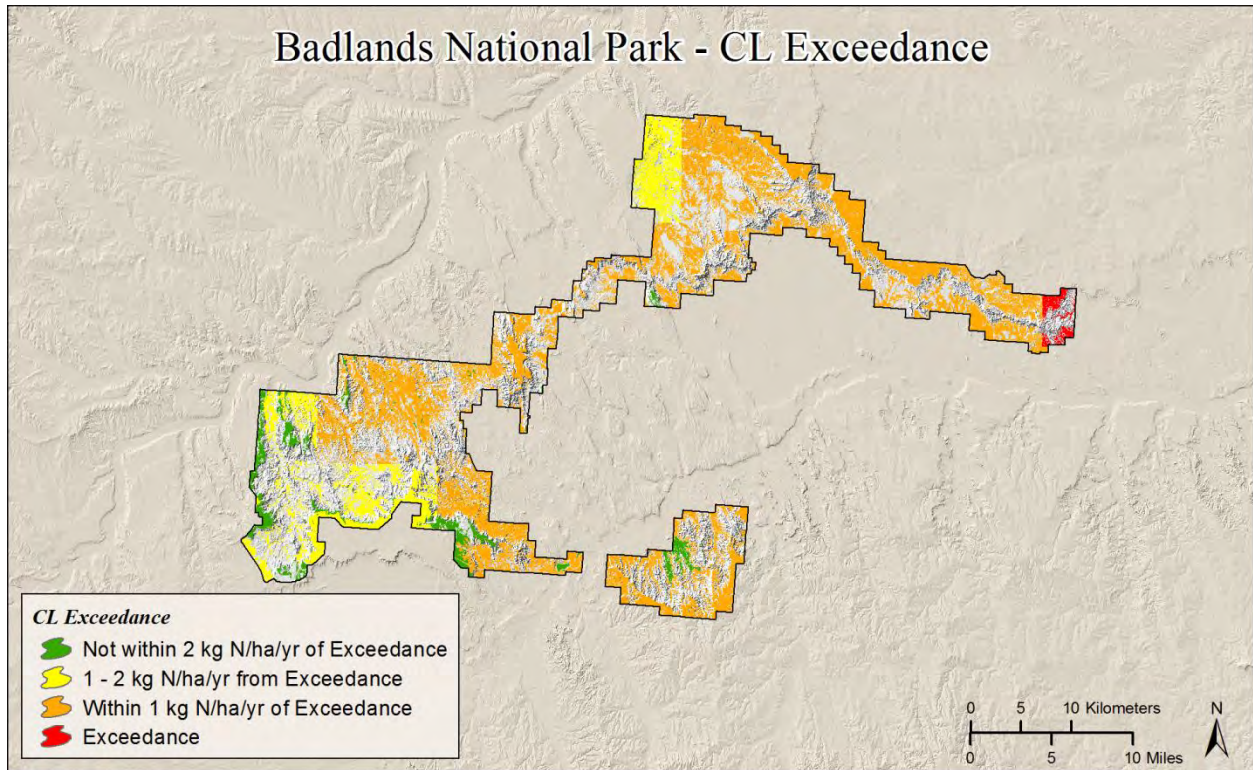
Map 47. Locations of vegetation types in Badlands National Park in exceedance or close to being in exceedance of the CL (5 kg N/ha/yr) to protect against nutrient-N enrichment. The area of sensitive vegetation is labeled red if total N deposition in year 2008 is in exceedance of the lower limit of the critical load (CL) estimated by Clark (2011). The remaining areas of sensitive vegetation are classified according to the extent to which total N deposition is below the CL. Areas showing the background color do not contain vegetation types identified by Clark (2011) as being highly sensitive to nutrient-N enrichment.
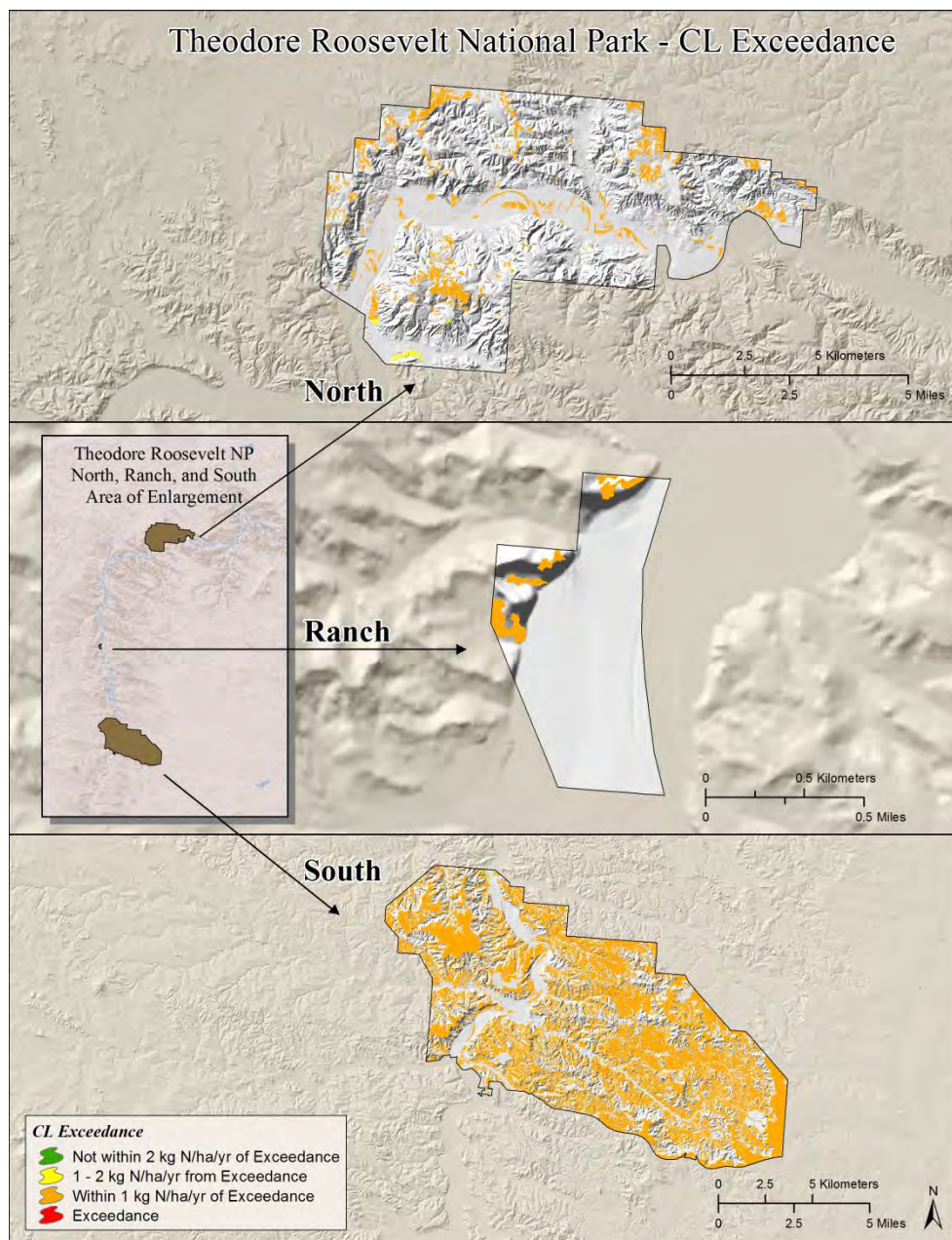
Map 48. Locations of vegetation types in Theodore Roosevelt National Park in exceedance or close to being in exceedance of the CL (5 kg N/ha/yr) to protect against nutrient-N enrichment. The area of sensitive vegetation is labeled red if total N deposition in year 2008 is in exceedance of the lower limit of the critical load (CL) estimated by Clark (2011). The remaining areas of sensitive vegetation are classified according to the extent to which total N deposition is below the CL. Areas showing the background color do not contain vegetation types identified by Clark (2011) as being highly sensitive to nutrient-N enrichment.
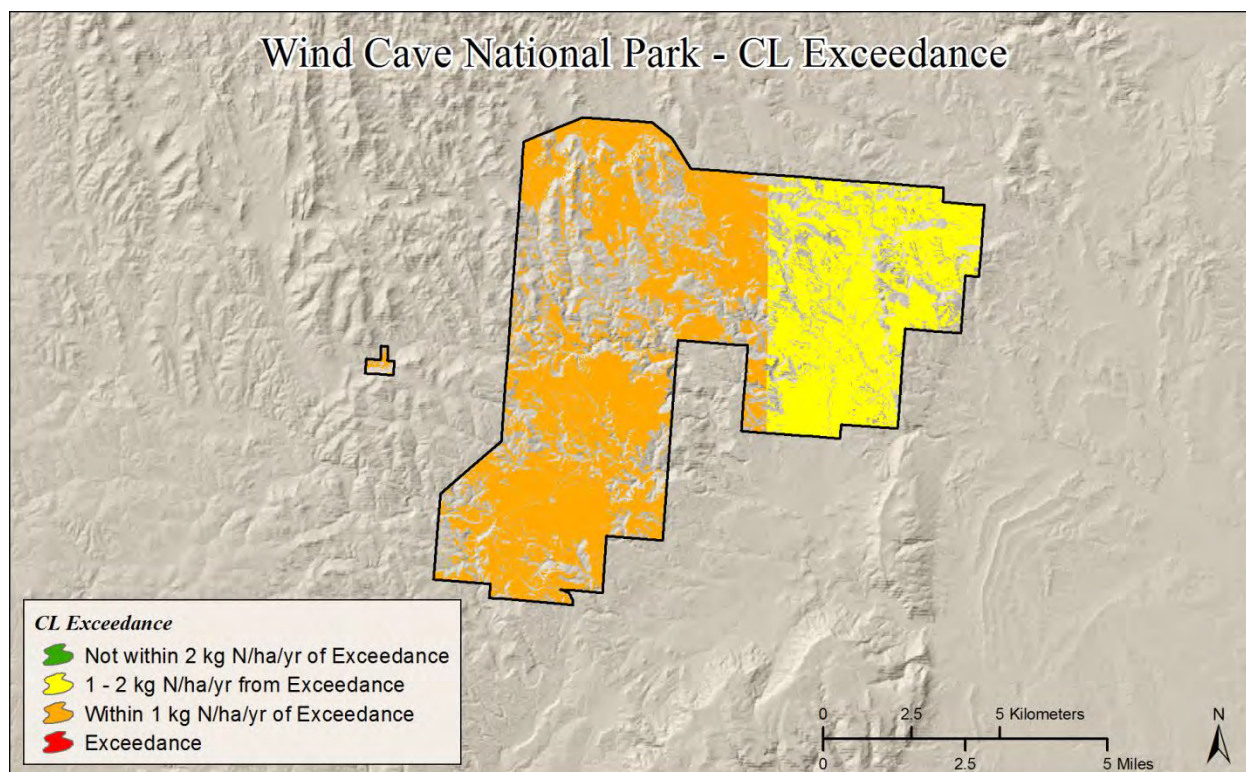
Map 49. Locations of vegetation types in Wind Cave National Park in exceedance or close to being in exceedance of the CL (5 kg N/ha/yr) to protect against nutrient-N enrichment. The area of sensitive vegetation is labeled red if total N deposition in year 2008 is in exceedance of the lower limit of the critical load (CL) estimated by Clark (2011). The remaining areas of sensitive vegetation are classified according to the extent to which total N deposition is below the CL. Areas showing the background color do not contain vegetation types identified by Clark (2011) as being highly sensitive to nutrient-N enrichment.

Literature Cited

Aber, J.D., C.L. Goodale, S.V. Ollinger, M.-L. Smith, A.H. Magill, M.E. Martin, R.A. Hallett, and J.L. Stoddard. 2003. Is nitrogen deposition altering the nitrogen status of northeastern forests? BioScience 53(4):375-389.

Aerts, R., J.T.A. Verhoeven, and D.F. Whigham. 1999. Plant-mediated controls on nutrient cycling in temperate fens and bogs. Ecology 80(7):2170-2181.

Aerts, R., F. Berendse, H. De Caluwe, and M. Schmits. 1990. Competition in heathland along an perimental gradient of nutrient availability. Oikos 57:310-318.

Aldous, A.R. 2002. Nitrogen retention by Sphagnum mosses: responses to atmospheric nitrogen deposition and drought. Can. J. Bot. 80(7):721-731.

Allen, E.B. and L.H. Geiser. 2011. North American Deserts. *In* L.H. Pardo, M.J. Robin-Abbott and C.T. Driscoll (Eds.). Assessment of Nitrogen Deposition Effects and Empirical Critical Loads of Nitrogen for Ecoregions of the United States. General Technical Report NRS-80. U.S. Forest Service, Newtown Square, PA. pp. 133-142.

Allen, E.B., L.E. Rao, and R.J. Steers. 2009. Impacts of atmospheric nitrogen deposition on vegetation and soils at Joshua Tree National Park. *In* R.H. Webb, J.S. Fenstermaker, J.S. Heaton, D.L. Hughson, E.V. McDonald and D.M. Miller (Eds.). The Mojave Desert: Ecosystem Processes and Sustainability. University of Nevada Press, Las Vegas. pp. 78-100.

Avis, P.G., D.J. McLaughlin, B.C. Dentinger, and P.B. Reich. 2003. Long-term increases in nitrogen supply alters above and belowground ectomycorrhizal communities and increases dominance of *Russula* species. New Phytol. 160(1):239-253.

Baer, S.G., J.M. Blair, S.L. Collins, and A.K. Knapp. 2003. Soil resources regulate productivity and diversity in newly established tallgrass prairie. Ecology 84:724-735.

Báez, S., J. Fargione, D.I. Moore, S.L. Collins, and J.R. Gosz. 2007. Atmospheric nitrogen deposition in the northern Chihuahuan desert: temporal trends and potential consequences. J. Arid Environ. 68:640-651.

Barker, D.H., C. Vanier, E. Naumburg, T.N. Charlet, K.M. Nielsen, B.A. Newingham, and S.D. Smith. 2006. Enhanced monsoon precipitation and nitrogen deposition affect leaf traits and photosynthesis differently in spring and summer in the desert shrub *Larrea tridentata*. New Phytol. 169:799-808.

Bedford, B.L., M.R. Walbridge, and A. Aldous. 1999. Patterns in nutrient availability and plant diversity of temperature North American wetlands. Ecol. Soc. Am. 80(7):2151-2169.

Belnap, J. 1991. Sensitivity of desert cryptogams to air pollutants: soil crusts and rock lichens. Tech. Rep. NPS/NRAQD/NRTR-91/02. *In* D. Mangis, J. Baron and K. Stolte (Eds.). Acid Rain and Air Pollution in Desert Park Areas. U.S. Department of the Interior, National Park Service, Washington, DC. pp. 112-119.

Belnap, J., S.L. Phillips, S. Flint, J. Money, and M. Caldwell. 2008. Global change and biological soil crusts: effects of ultraviolet augmentation under altered precipitation regimes and nitrogen additions. Glob. Change Biol. 14:670-686.

Berendse, F., N. van Breemen, H. Rydin, A. Buttler, M. Heijmans, M.R. Hoosbeek, J.A. Lee, A. Mitchell, T. Saarinen, H. Vassander, and B. Wallen. 2001. Raised atmospheric CO_2 levels and increased N deposition cause shifts in plant species composition and production in Sphagnum bogs. Glob. Change Biol. 7(5):591-598.

Bert, D., S.W. Leavitt, and J.-L. Dupouey. 1997. Variations of wood d^{12} C and water-use efficiency of *Abies alba* during the last century. Ecol. Soc. 78(1588-1596).

Bobbink, R., M. Hornung, and J.G.M. Roelofs. 1998. The effects of air-borne nitrogen pollutants on species diversity in natural and semi-natural European vegetation. J. Ecol. 86:717-738.

Bridgham, S.D., J. Pastor, C.A. McClaugherty, and C.J. Richardson. 1995. Nutrient-use efficiency: a litterfall index, a model, and a test along a nutrient-availability gradient in North Carolina peatlands. Amer. Natur. 145:1-21.

Brooks, M.L. 2003. Effects of increased soil nitrogen on the dominance of alien annual plants in the Mojave Desert. J. Appl. Ecol. 40:344-353.

Brooks, M.L. and J.R. Matchett. 2006. Spatial and temporal patterns of wildfires in the Mojave Desert, 1980-2004. Journal of Acid Environments 67:148-164.

Brooks, M.L., C.M. D'Antonio, D.M. Richardson, J.B. Grace, J.E. Keeley, J.M. DiTomaso, R.J. Hobbs, M. Pellant, and D. Pyke. 2004. Effects of invasive alien plants on fire regimes. BioScience 54:677-688.

Carreiro, M.M., R.L. Sinsabaugh, D.A. Repert, and D.F. Parkhurst. 2000. Microbial enzyme shifts explain litter decay responses to simulated nitrogen deposition. Ecology 81(9):2359-2365.

CEC (Commission for Environmental Cooperation). 1997. Ecological Regions of North America: Toward a Common Perspective. Commission for Environmental Cooperation, Montreal, Canada. http://www.cec.org/files/pdf/BIODIVERSITY/eco-eng_EN.pdf.

Clark, C.M. 2011. Great Plains. *In* L.H. Pardo, M.J. Robin-Abbott and C.T. Driscoll (Eds.). Assessment of Nitrogen Deposition Effects and Empirical Critical Loads of Nitrogen for Ecoregions of the United States. General Technical Report NRS-80. U.S. Forest Service, Newtown Square, PA. pp. 117-132.

Clark, C.M. and D. Tilman. 2008. Loss of plant species after chronic low-level nitrogen deposition to prairie grasslands. Nature 451:712-715.

Clark, C.M., S. Hobbie, R. Venterea, and D. Tilman. 2009. Long-lasting effects on N cycling 12 years after treatments cease despite minimal N retention. Glob. Change Biol. 15:1755-1766.

Clark, J.E., E.C. Hellgren, E.E. Jorgensen, S.J. Tunnell, D.M. Engle, and D.M. Leslie. 2003. Population dynamics of hispid cotton rats (*Sigmodon hispidus*) across a nitrogen-amended landscape. Can. J. Zool. 81:994-1003.

Cogan, D., H. Marriott, J.V. Loh, and M. Pucherelli. 1999. USGS-NPS Vegtation Mapping Program, Wind Cave National Park (Final Report). Technical Memorandum No. 8260-99-03. U.S. Bureau of Reclamation Technical Service Center, Denver, CO.

Dighton, J., A.R. Tuininga, D.M. Gray, R.E. Huskins, and T. Belton. 2004. Impacts of atmospheric deposition on New Jersey pine barrens forest soils and communities of ectomycorrhizae. For. Ecol. Manage. 201(1):131-144.

Drenovsky, R.H. and J.H. Richards. 2005. Nitrogen addition increases fecundity in the desert shrub *Sarcobatus vermiculatus*. Oecologia 143:349-356.

ENVIRON International Corporation. 2012. Western Regional Air Partnership (WRAP) West-wide Jump-start Air Quality Modeling Study (WestJumpAQMS) Draft Final Modeling Protocol. Prepared for Western Regional Air Partnership, CIRA, Colorado State University. Novato, CA.

Federal Geographic Data Committee (FGDC). 1996. FGDC Vegetation Classification and Information Standards. Reston, VA.

Fenn, M.E. and E.B. Allen. 2011. Southern semi-arid highlands. *In* L.H. Pardo, M.J. Robin-Abbott and C.T. Driscoll (Eds.). Assessment of Nitrogen Deposition Effects and Empirical Critical Loads of Nitrogen for Ecoregions of the United States. General Technical Report NRS-80. U.S. Forest Service, Newtown Square, PA. pp. 171-174.

Fenn, M.E., J.S. Baron, E.B. Allen, H.M. Rueth, K.R. Nydick, L. Geiser, W.D. Bowman, J.O. Sickman, T. Meixner, D.W. Johnson, and P. Neitlich. 2003. Ecological effects of nitrogen deposition in the western United States. BioScience 53(4):404-420.

Geiser, L.H., A.R. Ingersoll, A. Bytnerowicz, and S.A. Copeland. 2008. Evidence of enhanced atmospheric ammoniacal nitrogen in Hells Canyon National Recreation Area: Implications for natural and cultural resources. J. Air Waste Manage. Assoc. 58:1223-1234.

Geiser, L.H., S.E. Jovan, D.A. Glavich, and M.K. Porter. 2010. Lichen-based critical loads for atmospheric nitrogen deposition in western Oregon and Washington forests, USA. Environ. Pollut. 158:2412-2421.

Greaver, T., L. Liu, and R. Bobbink. 2011. Wetlands. *In* L.H. Pardo, M.J. Robin-Abbott and C.T. Driscoll (Eds.). Assessment of Nitrogen Deposition Effects and Empirical Critical Loads of Nitrogen for Ecoregions of the United States. General Technical Report NRS-80. U.S. Forest Service, Newtown Square, PA.

Gutierrez, J.R. and W.G. Whitford. 1987. Chihuahuan Desert annuals - importance of water and nitrogen. Ecology 68:2032-2045.

Haddad, N.M., J. Haarstad, and D. Tilman. 2000. The effects of long-term nitrogen loading on grassland insect communities. Oecologia 124:73-84.

Hautier, Y.R., P.A. Niklaus, and A. Hector. 2009. Competition for Light Causes Plant Biodiversity Loss After Eutrophication. Science 324:636-638.

Heil, G.W. and W.H. Diemont. 1983. Raised nutrient levels change heathland in grassland. Vegetation 53:113-120.

Heil, G.W. and M. Bruggink. 1987. Competition for nutrients between *Calluna vulgaris* (L.) Hull and *Molinia caerulea* (L.) Moench. Oecologia 73:105-108.

Hill, B.H., C.M. Elonen, T.M. Jicha, A.M. Cotter, A.S. Trebitz, and N.P. Danz. 2006. Sediment microbial enzyme activity as an indicator of nutrient limitation in Great Lakes coastal wetlands. Freshw. Biol. 51:1670-1683.

Hooper, D.U. and L. Johnson. 1999. Nitrogen limitation in dryland ecosystems: Response to geographical and temporal variation in precipitation. Biogeochemistry 46:247-293.

Hurd, T.M., A.R. Brach, and D.J. Raynal. 1998. Response of understory vegetation of Adirondack forests to nitrogen additions. Can. J. For. Res. 28:799-807.

Hutchinson, G.E. 1971. A Treatise on Limnology. Vol. 1. Geography, Physics, and Chemistry. John Wiley and Sons, New York.

Inouye, R.S. 2006. Effects of shrub removal and nitrogen addition on soil moisture in sagebrush steppe. J. Arid Environ. 65:604-618.

Jorgensen, E.E., S.M. Holub, P.M. Mayer, M.E. Gonsoulin, R.G. Silva, A.E. West, S.J. Tunnell, J.E. Clark, J.L. Parsons, D.M. Engle, E.C. Hellgren, J.D.H. Spears, C.E. Bulter, and D.M. Leslie, Jr. 2005. Ecosystem Stress from Chronic Exposure to Low Levels of Nitrate. EPA/600/R-05/087. U.S. Environmental Protection Agency, National Risk Management Research Laboratory, Washington, DC.

Kleb, H.R. and S.D. Wilson. 1997. Vegetation effects on soil resource heterogeneity in prairie and forest. Amer. Natur. 150:283-298.

Knapp, A.K. and M.D. Smith. 2001. Variation among biomes in temporal dynamics of aboveground primary production. Science 291:481-484.

Knapp, A.K., J.M. Briggs, D.C. Hartnett, and S.L. Collins (Eds.). 1998. Grassland Dynamics: Long-term Ecological Research in Tallgrass Prairie. Oxford University Press, New York. 386 pp.

Knops, J.M.H. and K. Reinhart. 2000. Specific leaf area along a nitrogen fertilization gradient. American Midland Naturalist 144:265-272.

Köchy, M. and S.D. Wilson. 2001. Nitrogen deposition and forest expansion in the Northern Great Plains. J. Ecol. 89:807-817.

Koerselman, W. 1989. Groundwater and surface water hydrology of a small groundwater-fed fen. Wetl. Ecol. Manage. 1:31-43.

Kooijman, A.M. and C. Bakker. 1994. The acidification capacity of wetland bryophytes as influenced by simulated clean and polluted rain. Aquat. Bot. 48(2):133-144.

Krupa, S.V. 2003. Effects of atmospheric ammonia (NH_3) on terrestrial vegetation: a review. Environ. Pollut. 124(179-221).

LANDFIRE. 2008. National Existing Vegetation Type Layer (Refresh 2008). Available from http://www.landfire.gov/NationalProductDescriptions21.php.

Lane, D.R. and H. BassiriRad. 2002. Differential responses of tallgrass prairie species to nitrogen loading and varying ratios of NO_3^- to NH_4^+. Funct. Plant Biol. 29:1227-1235.

Lauenroth, W.K., J.L. Dodd, and P.L. Simms. 1978. The effects of water- and nitrogen-induced stresses on plant community structure in a semiarid grassland. Oecologia 36:211-222.

Lilleskov, E.A. 2005. How do composition, structure, and function of mycorrhizal fungal communities respond to nitrogen deposition and ozone exposure? *In* J. Dighton, J.F. White and P. Oudemans (Eds.). The Fungal Community: its Organization and Role in the Ecosystem. Taylor and Francis, Boca Raton, FL. pp. 769-801.

Lilleskov, E.A., T.J. Fahey, and G.M. Lovett. 2001. Ectomycorrhizal fungal aboveground community change over an atmospheric nitrogen deposition gradient. Ecol. Appl. 11(2):397-410.

Lilleskov, E.A., T.J. Fahey, T.R. Horton, and G.M. Lovett. 2002. Belowground ectomycorrhizal fungal community change over a nitrogen deposition gradient in Alaska. Ecology 83(1):104-115.

Lilleskov, E.A., P.M. Wargo, K.A. Vogt, and D.J. Vogt. 2008. Mycorrhizal fungal community relationship to root nitrogen concentration over a regional atmospheric nitrogen deposition gradien in the northeastern US. 38:1260-1266.

Marsh, J.E. and T.H. Nash, III. 1979. Lichens in relation to the Four Corners Power Plant in New Mexico. Bryologist 82:20-28.

McCrackin, M.L., T.K. Harms, N.B. Grimm, S.J. Hall, and J.P. Kaye. 2008. Responses of soil microorganisms to resource availability in urban, desert soils. Biogeochemistry 87:143-155.

McNulty, S.G., J. Boggs, J.D. Aber, L. Rustad, and A. Magill. 2005. Red spruce ecosystem level changes following 14 years of chronic N fertilization. For. Ecol. Manage. 219:279-291.

Mitsch, W.J. and J.G. Gosselink. 2000. Wetlands. John Wiley & Sons, Inc., New York, NY. 936 pp.

Moore, D.R.J., P.A. Keddy, C.L. Gaudet, and I.C. Wisheu. 1989. Conservation of wetlands: do infertile wetlands deserve a higher priority? Biol. Cons. 47:203-217.

Moore, T., C. Blodau, J. Turenen, N. Roulet, and P.J.H. Richard. 2004. Patterns of nitrogen and sulfur accumulation and retention in ombrotrophic bogs, eastern Canada. Glob. Change Biol. 11:356-367.

Morrice, J.A., J.R. Kelly, A.S. Trebitz, A.M. Cotter, and M.L. Knuth. 2004. Temporal dynamics of nutrients (N and P) and hydrology in a Lake Superior coastal wetland. Journal of Great Lakes Research 30(Suppl 1):82-96.

Morris, J.T. 1991. Effects of nitrogen loading on wetland ecosystems with particular reference to atmospheric deposition. Ann. Rev. Ecol. Syst. 22:257-279.

National Parks Conservation Association (NPCA). 2013. National Parks and Hydraulic Fracturing. Balancing Energy Needs, Nature, and America's National Heritage. Washington, DC. http://www.npca.org/assets/pdf/Fracking_Report.pdf.

North Dakota Industrial Commission. 2012. Oil and Gas Production Report. North Dakota Indutrial Commission, Oil & Gas Division, Bismark, ND.

Ollinger, S.V., R.G. Lathrop, J.M. Ellis, J.D. Aber, G.M. Lovett, and S.E. Millham. 1993. A spatial model of atmospheric deposition for the northeastern US. Ecol. Appl. 3(3):459-472.

Pardo, L.H., M.J. Robin-Abbott, and C.T. Driscoll (Eds.). 2011a. Assessment of Nitrogen Deposition Effects and Empirical Critical Loads of Nitrogen for Ecoregions of the United States. General Technical Report NRS-80. U.S. Forest Service, Newtown Square, PA.

Pardo, L.H., C.L. Goodale, E.A. Lilleskov, and L.H. Geiser. 2011b. Northern forests. *In* L.H. Pardo, M.J. Robin-Abbott and C.T. Driscoll (Eds.). Assessment of Nitrogen Deposition Effects and Empirical Critical Loads of Nitrogen for Ecoregions of the United States. General Technical Report NRS-80. U.S. Forest Service, Newtown Square, PA. pp. 61-74.

Porter, M.K. 2007. Regional Modeling of nitrogen, sulfur and mercury atmospheric deposition in the Pacific Northwest. Masters thesis Thesis, Washington State University, Pullman, WA.

Rao, L.E., E.B. Allen, and T. Meixner. 2010. Risk-based determination of critical nitrogen deposition loads for fire spread in southern California deserts. Ecol. Appl. 20(5):1320-1335.

Rao, L.E., D.R. Parker, A. Bytnerowicz, and E.B. Allen. 2009. Nitrogen mineralization across an atmospheric nitrogen deposition gradient in southern California deserts. Journal of Acid Environments 73:920-930.

Redbo-Torstensson, P. 1994. The demographic consequences of nitrogen fertilization of a population of sundew, *Drosera rotundifolia*. Acta Botanica Neerlandica 43(2):175-188.

Rochefort, L., D.H. Vitt, and S.E. Bayley. 1990. Growth, production and decomposition dynamics of *Sphagnum* under natural and experimentally acidified conditions. Ecology 71(5):1986-2000.

Rose, C. and R.P. Axler. 1998. Uses of alkaline phosphatase activity in evaluating phytoplankton community phosphorous deficiency. Hydrobiologia 361:145-156.

Schlesinger, W.H., T.J. Ward, and J. Anderson. 2000. Nutrient losses in runoff from grassland and shrubland habitats in southern New Mexico: II. Field plots. Biogeochemistry 49:69-86.

Schwinning, S., B.I. Starr, N.J. Wojcik, M.E. Miller, J.E. Ehleringer, and R.S. Sanford. 2005. Effects of nitrogen deposition on an arid grassland in the Colorado Plateau cold desert. Rangeland Ecol. Manage. 58:565-574.

Seastedt, T.R. and A.K. Knapp. 1993. Consequences of nonequilibrium resource availability across multiple time scales: The transient maxima hypothesis. Amer. Natur. 141:621-633.

Shaver, G.R. and J.M. Melillo. 1984. Nutrient budgets of marsh plants - efficiency concepts and relation to availability. Ecology 655:1491-1510.

Shen, W., J. Wu, N.B. Grimm, and D. Hope. 2008. Effects of urbanization-induced environmental changes on ecosystem functioning in the Phoenix Metropolitan Region, USA. Ecosystems 11:138-155.

Stevens, C.J., N.B. Dise, O.J. Mountford, and D.J. Gowing. 2004. Impact of nitrogen deposition on the species richness of grasslands. Science 303:1876-1878.

Sundareshwar, P.V., J.T. Morris, E. Koepfler, and B. Fornwalt. 2003. Phosphorus limitation of coastal ecosystem processes. Science 299:563-565.

Thomas, R.Q., C.D. Canham, K.C. Weathers, and C.L. Goodale. 2010. Increased tree carbon storage in response to nitrogen deposition in the US. Nature Geoscience 3:13-17.

Throop, H.L. 2005. Nitrogen deposition and herbivory affect biomass production and allocation in an annual plant. Oikos 111:91-100.

Throop, H.L. and M.T. Lerdau. 2004. Effects of nitrogen deposition on insect herbivory: implications for community and ecosystem processes. Ecosystems 7:109-133.

Throop, H.L., E.A. Holland, W.J. Parton, D.S. Ojima, and C.A. Keough. 2004. Effects of nitrogen deposition and insect herbivory on patterns of ecosystem-level carbon and nitrogen dynamics: results from the CENTURY model. Glob. Change Biol. 2004(10):1092-1105.

Tilman, D. 1987. Secondary succession and the pattern of plant dominance along experimental nitrogen gradients. Ecol. Monogr. 57(3):189-214.

Tilman, D. 1993. Species richness of experimental productivity gradients: How important is colonization limitation. Ecology 74:2179-2191.

Tomassen, H.B., A.J. Smolders, J. Limpens, L.P. Lamers, and J.G. Roelofs. 2004. Expansion of invasive species on ombrotrophic bogs: desiccation or high N deposition? J. Appl. Ecol. 41(1):139-150.

Tomassen, H.B.M., A.J.P. Smolders, L.P.M. Lamers, and J.G.M. Roelofs. 2003. Stimulated growth of *Betula pubescens* and *Molinia caerulea* on ombrotrophic bogs: role of high levels of atmospheric nitrogen deposition. J. Ecol. 91:357-370.

Tonnesen, G., Z. Wang, M. Omary, and C.J. Chien. 2007. Assessment of nitrogen deposition: Modeling and habitat assessment. CEC-500-2005-032. California Energy Commission, PIER Energy-Related Environmental Research., Sacramento, CA. http://www.energy.ca.gov/2006publications/CEC-500-2006-032/CEC-500-2006-032.PDF.

U.S. Environmental Protection Agency. 1993. Air quality criteria for oxides of nitrogen. EPA/600/8-91/049aF-cF.3v. Office of Health and Environmental Assessment, Environmental Criteria and Assessment Office, Research Triangle Park, NC.

Van Diepen, L.T.A. 2008. The role and diversity of arbuscular mycorrhizal fungi in *Acer saccharum* dominated forest ecosystems under natural and N-amended conditions. PhD dissertation Thesis, Michigan Technological University, Houghton, MI.

van Diepen, L.T.A., E.A. Lilleskov, K.S. Pregitzer, and R.M. Miller. 2007. Decline of arbuscular mycorrhizal fungi in northern hardwood forests esposed to chronic nitrogen additions. New Phytol. 176:175-183.

van Diepen, L.T.A., E.A. Lilleskov, K.S. Pregitzer, and R.M. Miller. 2010. Simulated nitrogen deposition causes a decline of intra- and extraradical abundance of arbuscular mycorrhizal fungi and changes in microbial community structure in northern hardwood forests. Ecosystems 13:683-695.

Vitousek, P.M. and R.W. Howarth. 1991. Nitrogen limitation on land and in the sea: how can it occur? Biogeochemistry 13:87-115.

Vitt, D.H., K. Wieder, L.A. Halsey, and M. Turetsky. 2003. Response of *Sphagnum fuscum* to nitrogen deposition: a case study of ombrogenous peatlands in Alberta, Canada. Bryologist 106(2):235-245.

Von Loh, J., D. Cogan, D. Faber-Langendoen, D.W. Crawford, and M.J. Pucherelli. 1999. USGS-NPS Vegetation Mapping Program, Badlands National Park, South Dakota. U.S. Department of the Interior, Bureau of Reclamation's Remote Sensing and GIS Group, Denver, CO.

Von Loh, J., D. Cogan, J. Butler, D. Faber-Langendoen, D. Crawford, and M.J. Pucherelli. 2000. USGS-NPS Vegetation Mapping Program, Theodore Roosevelt National Park, North Dakota. U.S. Department of the Interior, Bureau of Reclamation's Remote Sensing and GIS Group, Denver, CO.

Wedin, D. and D. Tilman. 1996. Influence of nitrogen loading and species composition on the carbon balance of grasslands. Science 274:1720-1723.

Wilson, S.D. and D. Tillman. 1991. Components of plant competition along an experiment gradient of nitrogen availability. Ecology 72:1050-1065.

Appendix A: Compilation of Emissions Data

The NEI database includes NO_x emissions from permitted sources. However, the numerous individual small wells associated with oil and gas exploration and production do not require permits for operation. Emissions associated with these sources are generally not included in the NEI database. The WestJumpAQMS is a multifaceted effort to develop county-level emissions estimates by source type and to conduct air quality modeling in the western United States (Western Regional Air Partnership [WRAP] 2013). The study is managed by the Western Governors' Association as part of WRAP. One of the goals of the study was to develop a comprehensive emissions inventory which includes both permitted and unpermitted emissions related to oil and gas exploration and production. The most recent effort was designed to reflect year 2008 emissions and air quality conditions. Emissions data compiled by the WestJumpAQMS for 2008 were used in this analysis.

The WestJumpAQMS emissions database was developed in stages, based on the geographic region and the type of information available, including:

1. Emissions projections to 2008, using the 2006 baseline WRAP Phase III project inventories for the Rocky Mountain region, including the Denver-Julesburg (CO), Piceance (CO), Uinta (UT), North San Juan (CO), South San Juan (NM), Wind River (WY), Powder River (WY), Greater Green River (WY), and Williston (MT and ND, pending) basins,
2. Development of an independent 2008 Permian Basin (NM and TX) emissions Inventory,
3. Incorporation of emissions reported by states to EPA's 2008 NEI for the remainder of the United States.

Prior to the WestJumpAQMS, WRAP developed an emissions inventory of all criteria air pollutants (CAP) associated with oil and gas exploration and production activities for the year 2006 (WRAP Phase III; http://www.wrapair2.org/PhaseIII.aspx). The baseline WRAP Phase III oil and gas inventory for 2006 represented the most comprehensive and complete oil and gas inventory available for the WRAP study region at the time of the analyses reported here. It covers portions of Colorado, Montana, New Mexico, North Dakota, Utah and Wyoming (Map A1). Emissions projections for 2008 have been constructed for much of the WestJumpAQMS region. The WestJumpAQMS updated the 2006 WRAP Phase III emissions inventory to reflect 2008 emissions based on changes over that two-year period in oil and gas exploration and production activity and emissions controls. These 2008 estimates are used in the analyses reported here.

In addition to the updates made to the 2006 inventory, the WestJumpAQMS 2008 database included an emissions inventory for the Permian Basin of southeastern New Mexico and western Texas developed using the WRAP Phase III process. Emissions estimates for the years since 2008 have not been generated, although efforts are in progress to compile emissions data for 2011 for the Intermountain West through the Three State Air Quality Study and a WRAP study focused on the Williston and Great Plains basins. The Three State Study 2011 emissions inventories are expected to be completed by fall 2013. Similar emissions inventories are planned for the Williston Basin of North Dakota, South Dakota, and Montana and the Great Plains Basin of Montana. These inventories were not completed at the time this report was compiled.

Oil and gas emissions reported here were based on version 2.0 of the 2008 NEI database for areas not covered by the Permian Basin and other WRAP Phase III Basins included in the WestJumpAQMS. The oil and gas development-related emissions associated with activities such as transmission, distribution, storage and/or refining were categorized as other area and point source types and are not included in the oil and gas exploration and production figures reported here. Emissions of NO_x from all sources, not just those related to oil and gas development activity, were also obtained from the NEI to generate estimates of total emissions for each county and the percent of total emissions contributed by oil and gas exploration and production. A map of existing oil and gas development well locations as of 2006 is given by ENVIRON (2012).

NH_3 emissions associated with oil and gas development were not explicitly included in the either the WRAP Phase III or WestJumpAQMS study. However, NH_3 emissions reported to the NEI database were included in the deposition modeling. As a result, CAMx estimates of total N deposition used in this study include both oxidized and reduced forms of N but do not include any effort by WestJump to update NH_3 emissions data to account for additional emissions from numerous small sources.
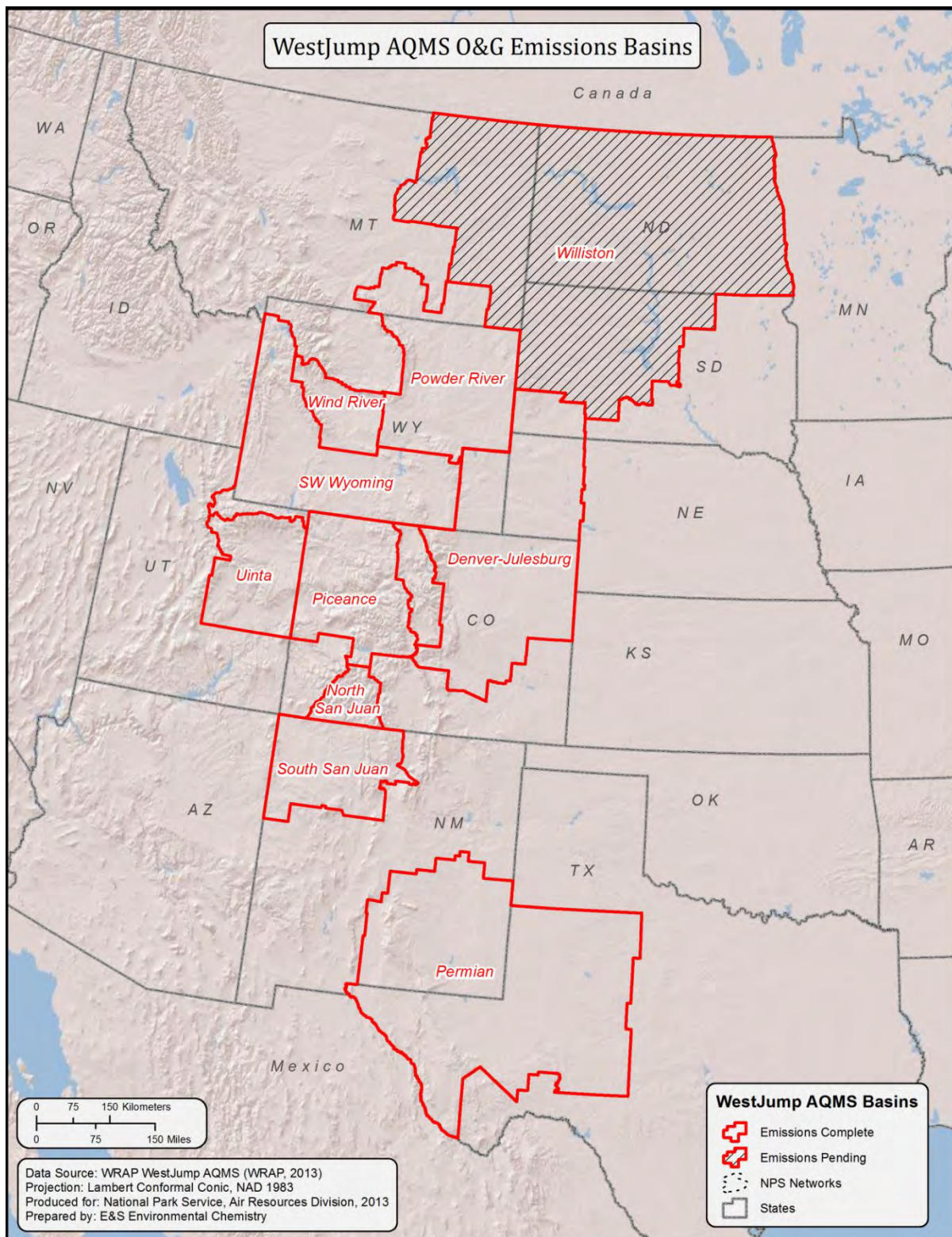
Map A1. Basins included in the WestJump effort.

# Appendix B: Overview of Nitrogen Emissions and Deposition

Emissions of N into the atmosphere throughout the conterminous United States from oil and gas development for the year 2008 are shown on Map B1. These emissions estimates were compiled for the most recent year for which data were available at the time of preparation of this report. Estimated oil and gas emissions for 2008 were especially high (>5,000 tons per year [tpy]) at scattered county locations in a band from the Northern Great Plains to the Texas Gulf Coast.

Existing data on oil and gas emissions that have the potential to adversely impact park resources within the study area are incomplete. There are several reasons for this. First, the NEI database includes data for some, but not all, small emissions sources. Because individual oil and gas wells tend to be small, many are not included in the database. The same may be true for small agricultural operations. Second, the recently developed WestJump database does include many of these small sources, but the spatial extent of the WestJump study is limited and the most recent year for which WestJump data are available at the time of this writing is 2008. Comparable estimates of NO_x emissions including recent oil and gas emissions data are not yet available for the eastern United States. Third, oil and gas development is now largely based on fracking, which constitutes about 90% of domestic oil and gas development in the United States (NPCA 2013). Technological developments that allow fracking have opened vast reserves of previously untapped resources, including energy development in shale plays throughout large portions of the United States. Use of this extraction method has increased at a rapid pace since 2008, and is projected to continue to increase. Thus, the emissions estimates for 2008 reported here may be lower than the current emissions levels at many locations. Fourth, fracking operations commonly occur at remote rural locations, sometimes in close proximity to national parks and monuments. Because oil and gas development has increased rapidly during the past five years, monitoring of NO_x emissions, N deposition, and associated O_3 formation and exposure has not kept pace. The environmental effects of this rapidly changing energy development technology remain largely unknown. Similarly, interactions with ongoing emissions increases and reductions from other source types are uncertain.
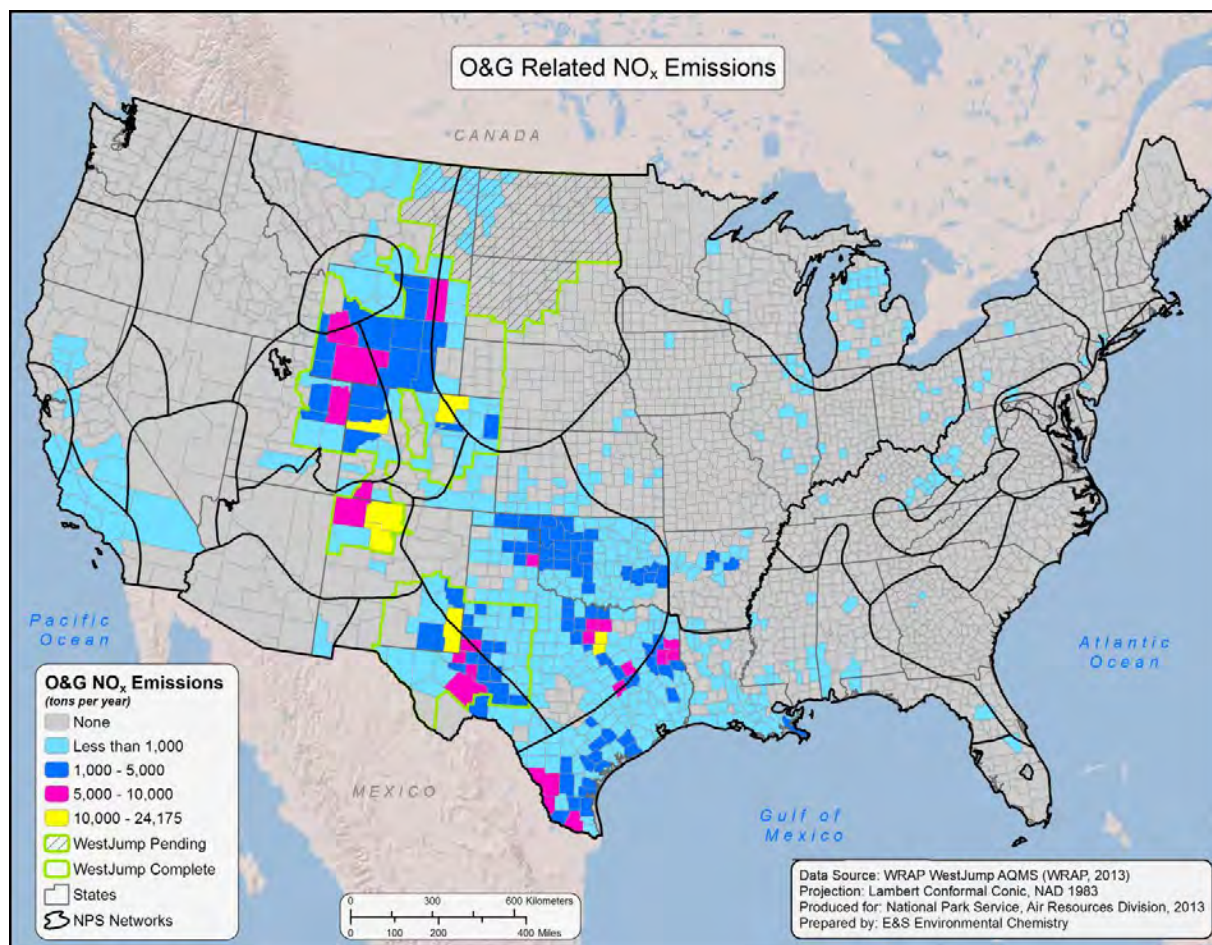
Map B1. Estimated oil and gas NOx emissions by county throughout the conterminous United States for the year 2008, which is the most recent year for which data are available from WestJump. Note that oil and gas emissions levels in some areas have increased dramatically in more recent years and are expected to continue to increase in the future. Note that detailed oil and gas emissions inventories comparable to the WestJump effort have not been developed for eastern states.

Estimated total wet plus dry N deposition across the regions included in this study are generally relatively low, compared to some other portions of the United States, including southern California and much of the eastern United States (Map B2). Nevertheless, as shown in this report, N deposition at the locations of many national parks included in this analysis is elevated in comparison with empirically determined CL estimates compiled by Pardo et al. (2011a).
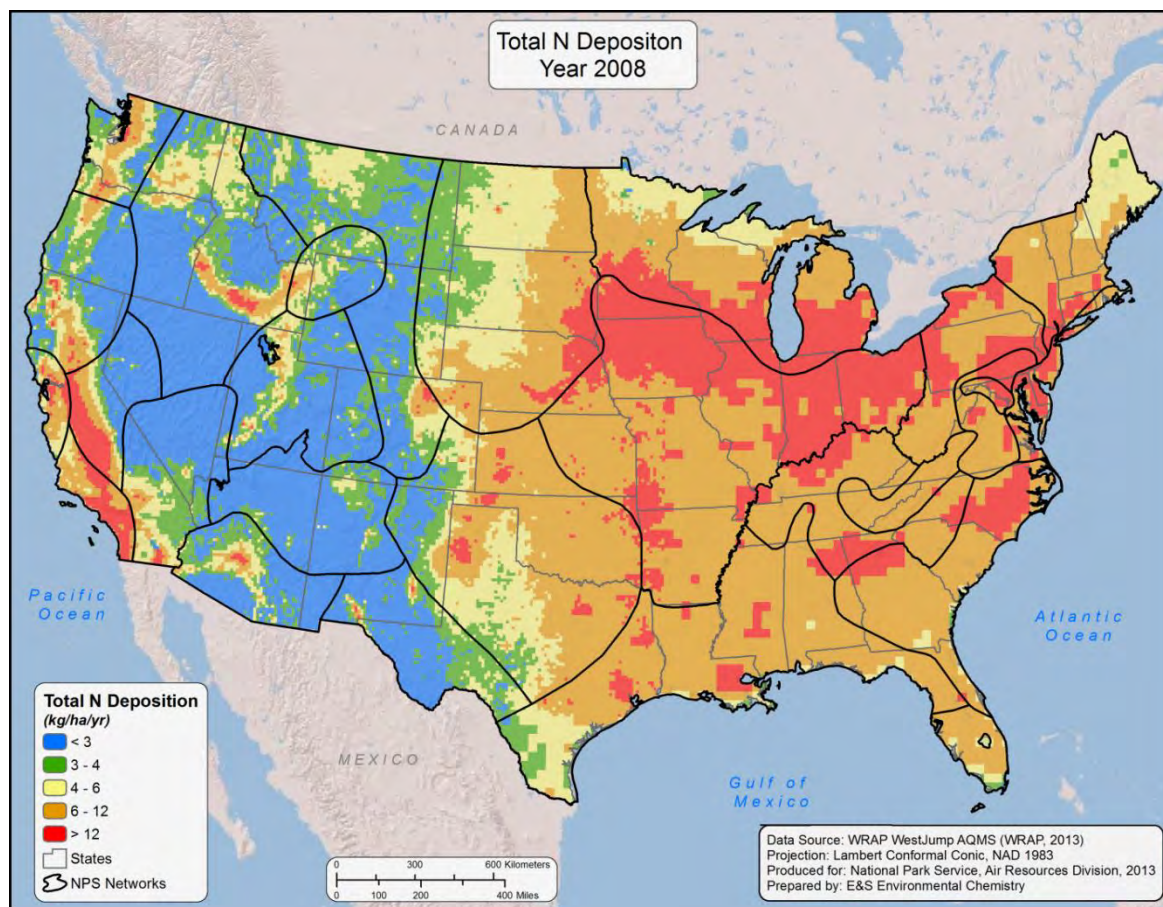
Map B2. Estimated total wet plus dry atmospheric deposition of N throughout the conterminous United States, based on CAMx model estimates for the year 2008. Data is presented at 12 km grid resolution for western states and 36 grid resolution for eastern US.

Natural Resource Stewardship and Science
1201 Oakridge Drive, Suite 150
Fort Collins, CO 80525

www.nature.nps.gov

EXPERIENCE YOUR AMERICA ™